Hydrogeologic Investigations of the Taylor Pass Landfill, Blenheim, New Zealand

VOLUME ONE

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ABSTRACT

An engineering geological and hydrogeological investigation has been carried out in the Taylor Pass Landfill area, some 2.5km south-south-west from the township of Blenheim. The primary aims of the study have been to identify the nature and extent of any leachate plume that is being generated from the now-closed landfill site, and to specifically determine the relative contributions (if any) to leachate generation from surface infiltration and groundwater seepage inflows. Investigation methods have included site mapping and trenching, TEM and electrical resistivity surveys, installation of a monitoring network of bores to depths up to 30m, and routine groundwater chemistry sampling and analysis. Leachate plume delineation has relied primarily on water chemistry data constrained by the hydrogeological model determined for the site, as geophysical techniques proved unsatisfactory due to background noise.

The Taylor Pass Landfill is located on the surface of the Taylor Fan, mostly within a former aggregate quarry, and was operational from 1976 to 1995 (although some offal waste is still being disposed of at this site pending remediation of the new Blenheim landfill at Blue Gums). The site hydrogeology is complex, with fan gravels, sands, silts and clays of the Rapaura Formation (<14,000 years BP) underlying the landfill to estimated depths of 15-25m, and a complex interplay of channel migration and overbank deposition being indicated by the available borehole logs and site mapping. Hydraulic conductivities and transmissivities range respectively from 3×10^{-4} to 8×10^{-4} m/s and 530 to 1400, with the higher values relating to the channel deposits within the various units of a prograding and migrating alluvial fan. Capping materials for the landfill are variable, although most are loess silts from the nearby Wither Hills, and laboratory-determined permeabilities are typically less than 10^{-7} m/s with some materials up to m/s.

Monitoring bores both up-gradient and down-gradient from the Taylor Pass Landfill reveal a relatively saline natural groundwater associated with the low summer flows and high evaporation rates of the southern part of the Wairau Valley. Groundwaters of the Wairau Aquifer occur some 1-1.5km down-gradient of the Landfill site, and are clearly of much higher quality and entirely suited to domestic use, with low ion concentrations, and low to undetectable metal levels. Leachate has been identified by characteristic high bicarbonate, ammonia, manganese, potassium, and calcium, and marginally elevated arsenic, sodium, chloride and sulphate, but interpretation of the plume is complicated by the presence of an older landfill at Brayshaw Park some 1.5m down-gradient from the Taylor Pass Landfill and the fact that there is only limited data available from deeper (>25m) monitoring wells.

It is concluded that the leachate plume from the TP Landfill extends as far north as the Wairau Aquifer, where the substantially higher flows rapidly attenuate the plume without impacting on down-gradient users. A zone of mixing of Taylor Fan and Wairau-derived groundwaters occurs for some 500m up-gradient from the mapped boundary near New Renwick and Alabama Roads, and is considered to account for the apparent plume termination in this vicinity. Additional capping of the Taylor Pass Landfill is not thought to be justified given the interpreted extent of the leachate plume and its rapid attenuation, but further tree-planting and cessation of offal disposal is strongly recommended to reduce further the extent of leachate generation. The drilling of at least three monitoring bores through the centre of the Landfill is also recommended given the absence of reliable data on groundwater levels and fluctuations within the area of refuse.

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Now, bring on the next chapter...Panama!

Ps I think hell just froze over!

Chapter 1

Introduction

1.1 Background

Over recent decades, public and industry alike have become increasingly aware of the impact of human activity on the environment. Urban and industrial waste management and disposal have become particularly sensitive issues, with the potential impact on the quality of soil and water resources coming under scrutiny. With the increased demand upon natural resources, and the realisation of their vulnerability, has come a movement towards safer waste disposal ensuring the conservation of valuable natural resources.

While new sanitary landfill sites in New Zealand are progressively being constructed to strict environmental guidelines, many older unsanitary sites (abundant in both rural and residential New Zealand) are being recognised as threats to the quality of local water and soil resources. New Zealanders have amongst the highest levels of waste production per capita (Ministry for the Environment, 1992), yet less than 17% of 981 disposal sites surveyed nationwide in 1988 met minimum World Health Organisation (WHO) standards (Drury and Towle 1992). This comes as an immediate result of the "out of sight, out of mind" philosophy for waste disposal in the past where little or no appreciation was given to the effects of discarding waste materials into the environment. More recent integrated waste management philosophies are based on "reduce, reuse, recycle and recover" themes. Reduction of waste to landfills encourages both the protection, and the responsible and controlled utilisation, of natural resources. In New Zealand's foreseeable future, it is likely that landfills will remain an important component of waste management.

> *"Waste is composed of solid, liquid, and gaseous material which may be classed as hazardous, toxic, or inert. Waste is generated as a consequence of using and developing resources. Waste should be collected and managed in a way which avoids, remedies or mitigates adverse effects on the environment"* (from the Marlborough Regional Policy Statement, in Royds Consulting, 1994.)

This attitude employed by the Marlborough District Council with respect to past and present waste disposal has led to the identification of the Taylor Pass Landfill, near Blenheim, as a potential threat to the surrounding environment due to disposal of wastes in what is now considered an unacceptable fashion. Thus an investigation of the local geological setting and consequent hydrogeological processes in the Taylor Fan Area, and the geotechnical nature and hydrological aspects of the Taylor Pass Landfill, has been undertaken in order to determine the impact of the Taylor Pass Landfill on local groundwater quality.

The remainder of Chapter 1 discusses the history of the Taylor Pass Landfill site and introduces the basic hydrology and hydrogeology, vegetation and climate, giving an overview of the regional setting. Regional and local water uses that provide the impetus for this project are also discussed.

1.2 Regional setting

1.2.1 Location

South of the Marlborough Sounds and north of the Awatere Catchment, the Wairau Catchment runs in a north-easterly direction from the northern end of the main southern divide in the west to Cloudy Bay in the east. The catchment covers an area of 3,825 km^2, most of which lies to the south of the Wairau River itself (Figure 1.1). Blenheim, the largest settlement within the catchment, lies approximately 10 km from the Cloudy Bay coast at the foot of the Wither Hills, on the southern margin of the Wairau Plains. The Taylor Pass Landfill is located approximately 4 km south of Blenheim in the Taylor Fan area. Note that all grid references in this thesis are given with respect to NZMS 260 – Sheet P28 (Department of Lands and Survey, 1982). Note that all grid references in this thesis are with respect to NZMS 260 – Sheet P28.

1.2.2 Hydrology and Hydrogeology

An extensive aquifer system formed by a series of glacial and interglacial deposits underlies the Wairau Plains in a manner analogous to the Canterbury Plains Aquifer System (Thorpe, 1992). The major source of recharge is the Wairau River at the northern margin of the Plains. As a direct result, well yields in the northern fringe of the plains are substantially higher than those in the south. The Wairau River originates from precipitation in the upper Wairau Catchment and tributary rivers throughout its length. Tributary river flows with respect to individual catchment size decrease towards the coast, reflecting the rainfall pattern over the catchment.

Current work being carried out by the Marlborough District Council is aimed at identifying the nature of the aquifer systems in the southern tributary valleys and the connection between the main Wairau Aquifer and aquifers of the Woodbourne and Fairhall areas. The existence of an extensive regional aquifer system beneath the Wairau Plains, and a lack of demand for water in the Taylor

Fan area, have meant that little investigative work has been done on the Taylor Fan itself. The present investigation is effectively the first hydrogeological study to be carried out in the Lower Taylor Fan area, and a detailed discussion on the hydrology and hydrogeology is therefore presented in Chapter 4.

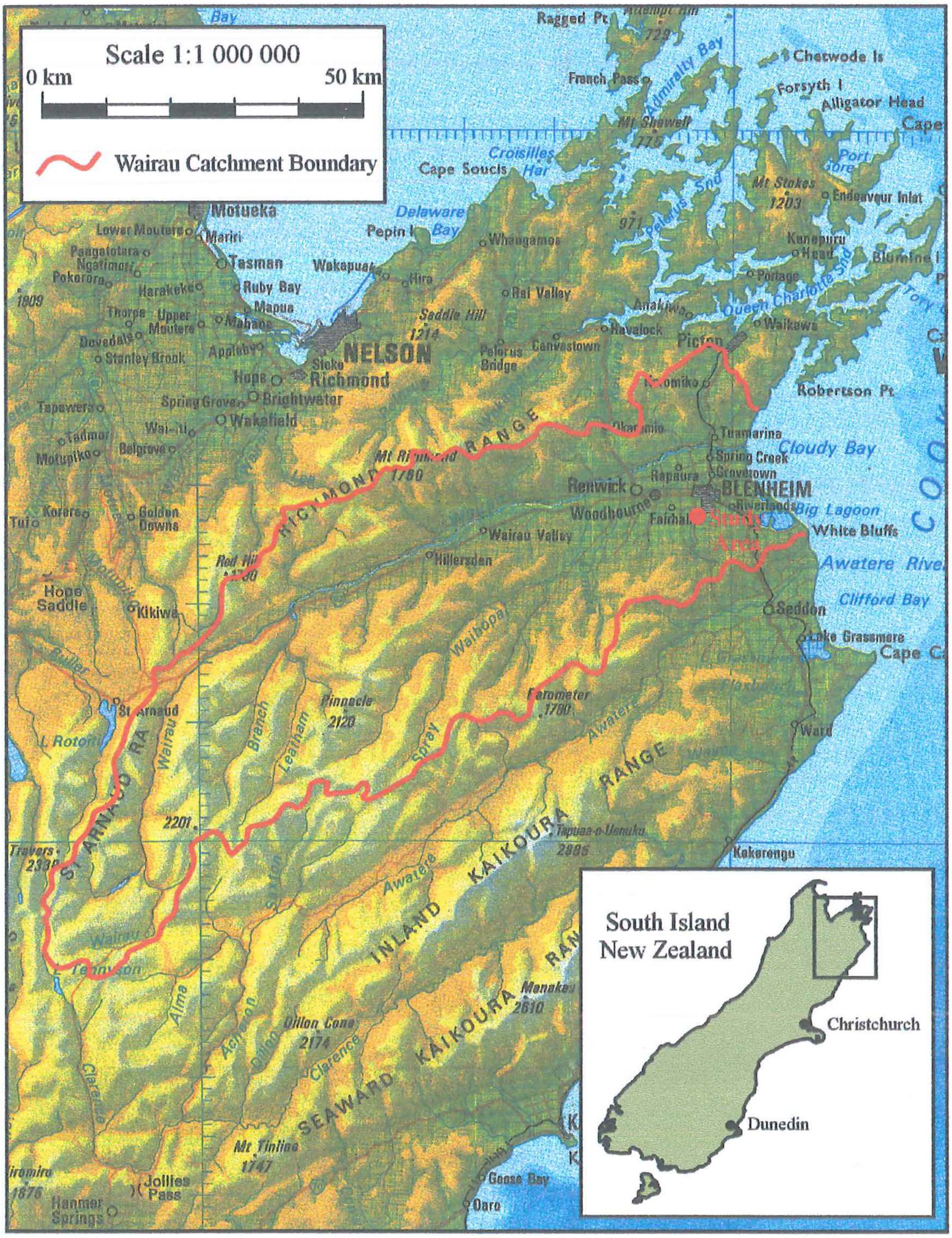


FIGURE 1.1: WAIRAU CATCHMENT LOCATION MAP

1.2.3 Climate

The geography of the mountain and valley system, sheltering effects of the North Island, and channelling of wind-streams through the Cook Strait, cause Marlborough to be amongst the driest regions of the South Island. A substantial rainfall gradient exists over the plains area in both west-east and north-south directions. Annual rainfall at the head of the Wairau River in the west is in excess of 4 metres, compared to less than 600 millimetres in the Cloudy Bay area to the east. Likewise the northern tributary valleys receive in the vicinity of 2 m of annual rainfall compared to only 800 millimetres in the Taylor River catchment (Figure 1.2).

Temperatures in the Blenheim area range over approximately 35°C, with mean monthly temperatures from 7-8°C in winter up to 18-19°C in summer. Blenheim boasts the greatest average hours of sunshine in New Zealand with 2,449 hours per year. The valley-sea geography controls wind conditions, leading to dominant easterly or westerly winds in Blenheim with an average wind speed of only 13 km/hr. Humidity ranges from 63% in the summer up to 84% in July. Frosts are recorded at an average rate of 32 days per year; the frequency of other meteorological phenomenon is shown in Table 1.1.

1.2.4 Vegetation and Land Use

Rae and Tozer (1990) give a good overview of vegetation history and trends in the Wairau Valley region, and in various ecological subsystems over the Wairau Plains are described. Prior to human influence, flax, raupo, toitoi and cabbage trees in a wetland environment occupied the lower Wairau Plains, south of the Wairau River, to the Wither Hills. Blenheim was formerly named Beavertown due to the swampland environment encountered upon initial settlement. Within the wetland, patches of kahikatea, pukatea and swamp mire formed swamp forest patches. On drier land, communities of open forestland extended through the proximal Taylor and Fairhall River and other southern tributary fans, into the Wither and Vernon Hills and over much of the central Pains region. Tree brooms, kowhai, kanuka, matai, and totara trees within shrubland and short tussock grass typified northern facing slopes, with less common rainforest species on south facing slopes.

Station	Snow	Hail	Thunder	Gales	Fog
Blenheim Aerodrome	0.2	0.2	2	1.9	5.2
Blenheim	0.1	0.9	3.5	2.2	1.9
Wither Hills	0.2	-	-	0.2	0.6

TABLE 1.1: OCCURRENCE OF METEOROLOGICAL PHENOMENA - DAYS PER YEAR. (FROM RAE, 1987).

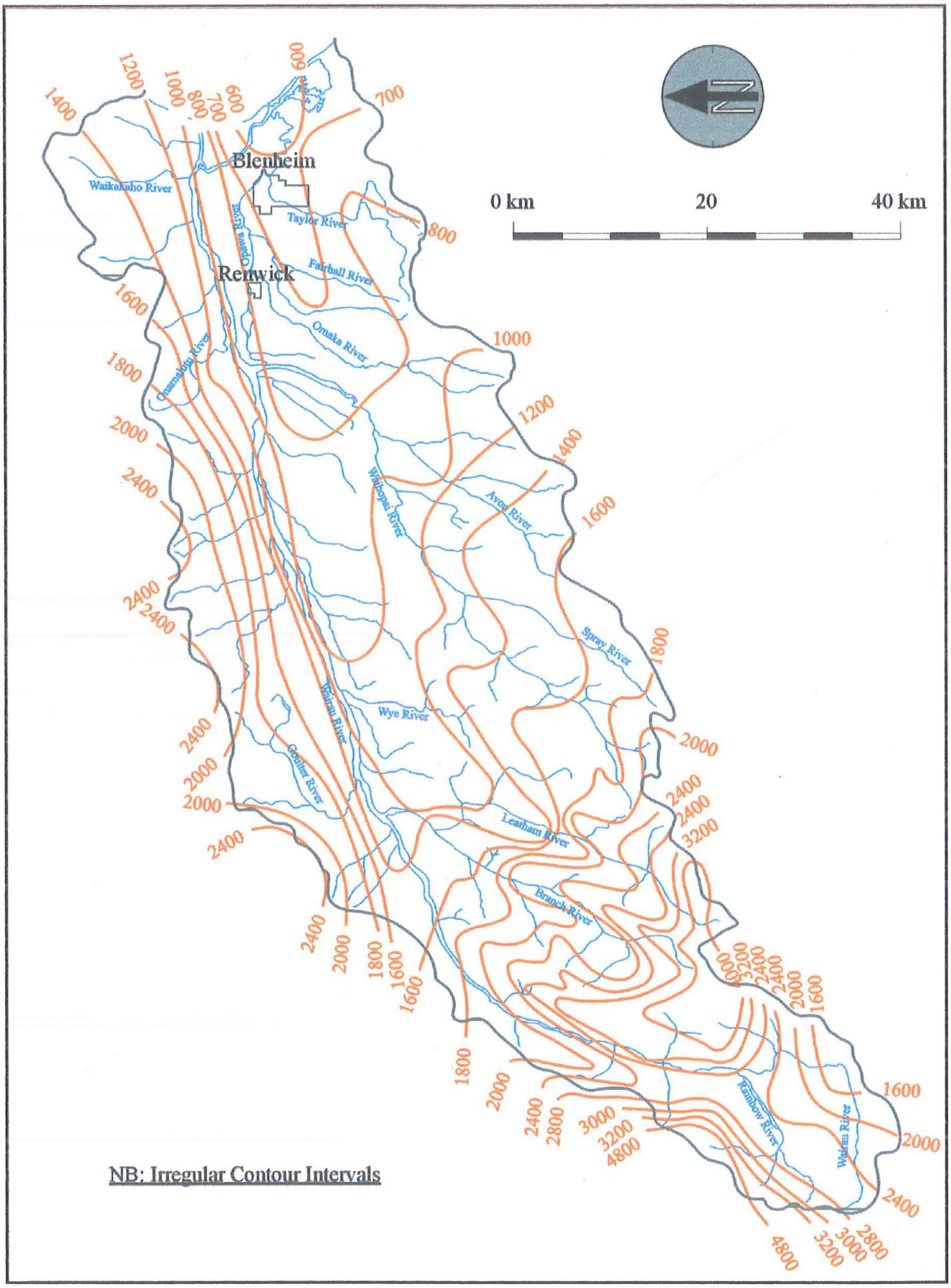


FIGURE 1.2: ISOHYETAL MAP OF THE WAIRAU CATCHMENT. RAINFALL IN MM/YEAR (FROM RAE, 1987).

McGlone (1983, in Rae and Tozer, 1990) suggests that much of the initial deforestation in the south bank area was due to accidental and deliberate fires between 800 and 600 years B.P. Maori and early European inhabitants no doubt also prevented successful reforestation. European cultural and industrial influences saw a major change in vegetation as land use expanded to include much pastoral farming, forest clearance and flax milling. In the Wither Hills area, deforestation caused significant erosion problems on the loess-covered slopes, supplying substantial volumes of fine grained sediment to local alluvial systems.

Increased settlement in the area encouraged the initiation of significant wetland drainage and flood control works. Agricultural and pastoral activities intensified following river control works in the late 1800's. Presently the Wairau Plains is known for its diversity of viticultural, horticultural, agricultural and pastoral industries, which form the backbone of the local economy. The upper Taylor Fan area is utilised primarily for cattle and sheep farming; the distal fan area is predominantly industrial, merging into the residential outskirts of Blenheim to the northeast; whilst viticultural activity is slowly developing to the west of the Taylor River in the distal fan region.

1.3 Water use

1.3.1 Regional water use

Water is an important natural resource in the Wairau Plains area as described by Rae (1987). Surface waters are enjoyed by many for recreational pursuits, drawn upon for irrigation and rare stock watering, and for domestic purposes. The Waihopai and Branch Rivers are both host to hydroelectric stations completed in 1927 and 1983, respectively. Whilst the Branch Scheme is still operational, the Benhopai Station on the Waihopai River filled rapidly with gravels following construction and is no longer functional. Waters drawn from Spring Creek, north of Blenheim, are used for salmon farming purposes. Various watercourses are also sites for stormwater disposal and emergency agricultural, trade and municipal effluent discharge.

Commercially, groundwater remains the most important water resource, supplying the region's drinking waters and also utilized extensively for irrigation purposes for the Wairau Plain's widespread agricultural and horticultural activities. Demand on Wairau groundwater from viticultural and primary produce industries is especially high during the dry summer season, and seasonal depletion of groundwater resources acts as a hindrance to further resource development. Industrial and municipal supplies come from the groundwater system; a number of town supply wells are in the immediate surrounds of the Blenheim Township. More recent uses of groundwater resources are for frost control in the stone fruit industry and abstraction for salmon farming.

Viticultural and horticultural irrigation remains however the largest user of groundwaters across the Wairau Plains.

The Marlborough District Council commits substantial resources to groundwater investigations and management aimed at gaining a further understanding of the extensive Wairau Aquifer system and the demands placed on it by groundwater abstraction. Monitoring of abstraction rates and volumes of larger users is carried out to ensure sustainability of the Wairau's most important natural resource.

1.3.2 Lower Taylor Fan Water Use

Due to its proximity to Blenheim, the Taylor River is among the most popular of the southern tributaries for recreational pursuits; however surface waters in the Taylor River are seasonal and sporadic. Construction of the Taylor Dam in 1964-65 as a flood protection measure for the Blenheim Township has led to a permanent body of standing water upstream of the dam, which is commonly used for recreational purposes. Below the Taylor Dam, surface water in the Taylor River disappears and reappears within the gravels, until it is in a constantly flowing state between flood protection banks through the Blenheim Township. Taylor River surface water flows are generally unreliable, however, the farmers in the upper section of the Taylor Fan draw stock water from the river when possible or required.

Groundwaters in the Upper Taylor Fan are drawn on for both domestic use and stock watering purposes. Numerous shallow wells (< 25 meters in depth) and less common deeper wells access groundwater in the Taylor Fan area. Shallow domestic supply wells tapping the Taylor Fan system were in use throughout the Redwoodtown area prior to the installation of a reticulated town water supply. Most such Redwoodtown wells have since been abandoned (Royds Consulting, 1994), and domestic supply wells remain only in unreticulated areas (e.g. Green Lane). Groundwater extracted from the Taylor Fan system is restricted to that for domestic supply, stock, and minor irrigation purposes due to the relatively low and inconsistent yields compared to the neighbouring Wairau aquifer system (Chapter 4).

Town supply wells are located within the Wairau Aquifer, near the margin of the Taylor system. Water is extracted for town supplies from the confined Wairau aquifer, at depths of from 20-35 m. The closest town supply well to the Taylor Pass Landfill is located at Eltham Road (well number P28/1313), approximately 2 km to the north, and has the most potential for contamination from the Taylor Pass Landfill. The Eltham Road well is employed only as a back up well and thus remains largely unutilised.

1.4 Site history

1.4.1 Taylor Pass Landfill

Taylor Pass landfill lies on the true right of the Taylor River approximately 4 km southwest of Blenheim, New Zealand (Figure 1.3). Farmland owned by the Marlborough District Council (MDC) surrounds the southern and eastern margins; industrial premises occupy sites north of the landfill; and an area of river gravels lie to the west. The landfill covers a 23-hectare site, originally founded as a gravel extraction pit. A diversion channel formed during gravel extraction runs around the southern and western margins of the site, directing a spring fed stream around the landfill (Figure 1.4).

Filling at the site commenced in 1976 at the northern end of the landfill, continuing southwards, until closure in October 1996. From 1993, a compactor was introduced and used at the site. Thus refuse is compacted only at the southern end of the filling area. Prior to 1994 no regular monitoring of the site had been carried out and no measures had been taken to control environmental damage (Royds Consulting, 1994).

Current Utilisation

Some illegal dumping of rubbish at the Taylor Pass Landfill site still occurs on a very small scale. However, following closure of the Taylor Pass Landfill in October 1996, organised waste disposal activity moved to the Regional Blue Gums lined sanitary landfill approximately 2 km south of the Taylor Pass Landfill site (Figure. 1.3). Initial construction activities and leakage problems within a filling cell at the Blue Gums site, however, caused concern for the safe disposal of liquid wastes. Resource consent was thus granted to discharge liquid wastes into excavated pits within the Taylor Pass Landfill for a period of three years whilst Blue Gums site problems were rectified.

Liquid waste disposal at the Taylor Pass Landfill currently includes cheese factory sludge wastes, animal dung, sludges (e.g. winery sludges), some oil and hydrocarbons, and offal with a combined volume of approximately 20 m^3 per day. The New Zealand Ministry of Agriculture and Fisheries (MAF, 1991) suggest the possibility of elevated levels of spore-forming or other resistant organisms as a consequence of the disposal of animal wastes to landfills, however no specific cases have been confirmed. The liquid wastes obviously contribute greatly to the production of leachate, but excavated pits are all located within an area controlled by a leachate recovery system. Resource consent for the disposal of liquid wastes expires in 2001.

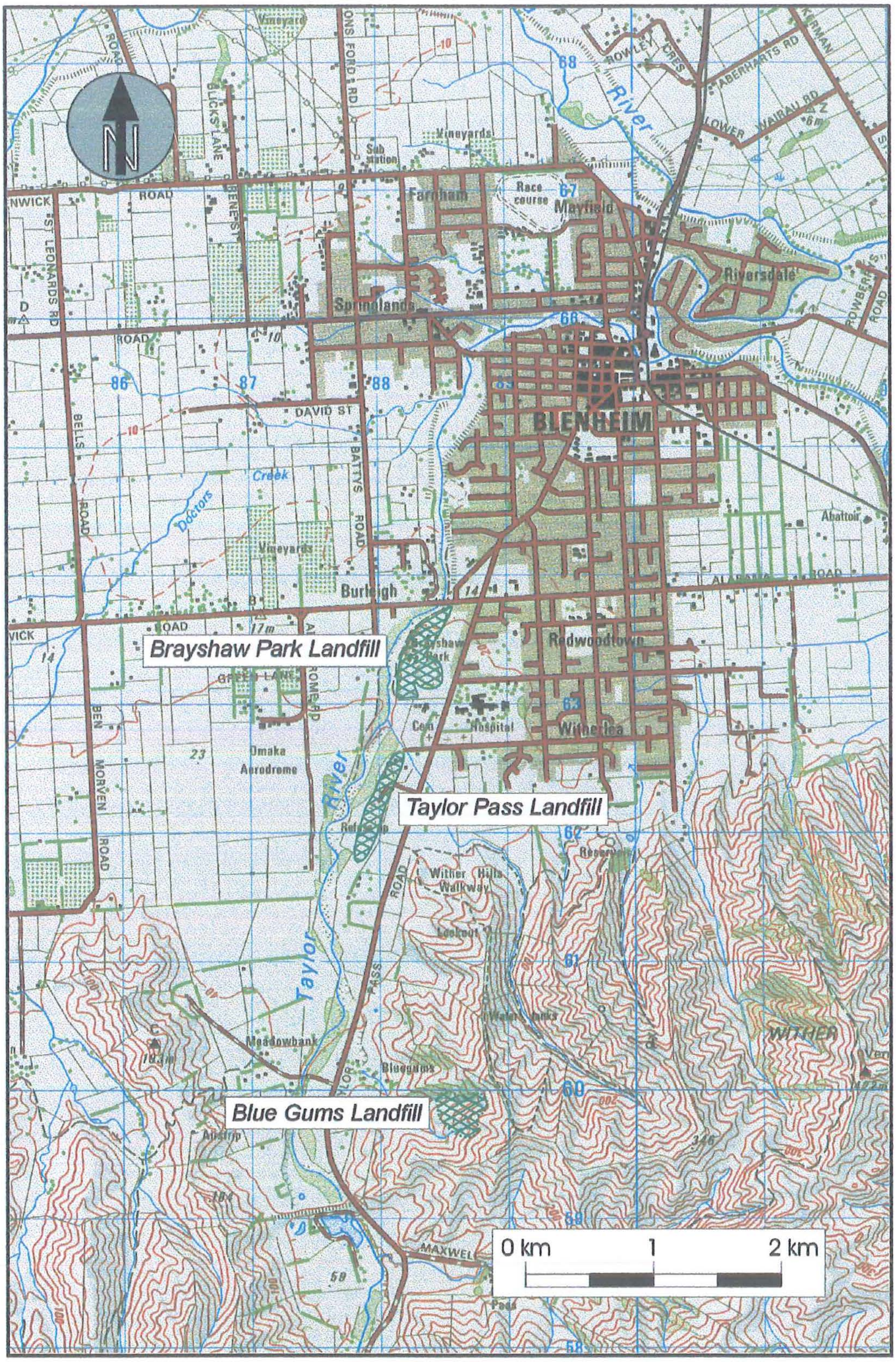


FIGURE 1.3: LOCATION MAP – TAYLOR PASS LANDFILL, BLENHEIM (NZMS 260 –SHEET P28).

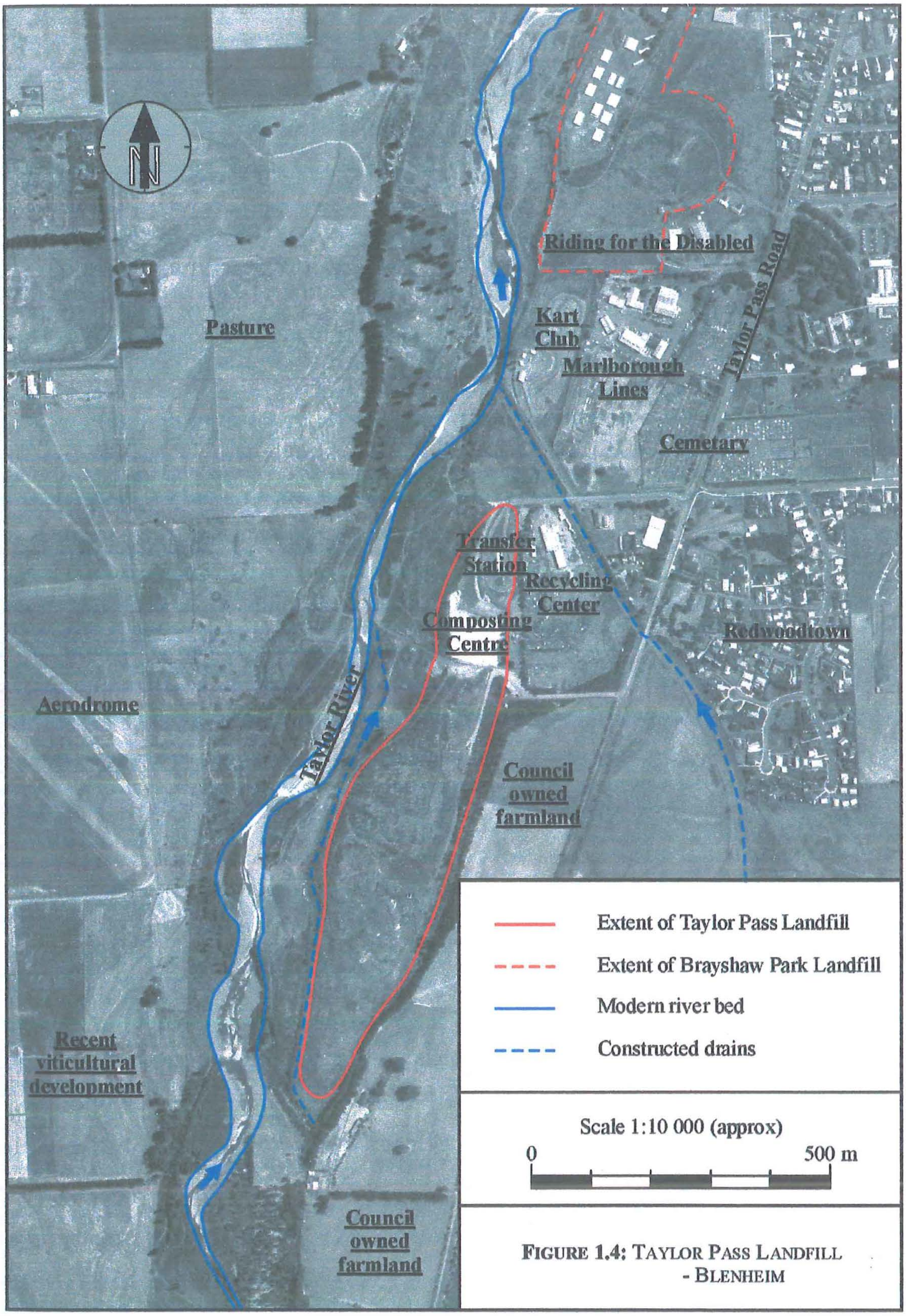


FIGURE 1.4: TAYLOR PASS LANDFILL - BLENHEIM

Closure Requirements

Closure conditions for the Taylor Pass Landfill are imposed under the Resource Management Act (1991) consent number U940852 to minimize environmental effects. An annual review required by consent conditions assesses post-closure progress and monitoring results. Closure conditions relevant to this investigation are summarised in Table 1.2, and relate to landfill capping and leachate monitoring and control. Further discussion of the requirements is in subsequent chapters. Capping requirements for resource consent are the main impetus for this investigation. Consent requires that the landfill be covered with a minimum of 450 mm of material with a permeability of no greater than 10^{-7} m/s. Also stipulated in resource consent conditions are minimum and maximum slope angles based on run-off and stability factors respectively. Finished slope angles, which were initially required to be between 1:20 and 1:4, have been relaxed to between 1:20 and 1:3 due to reconsideration of slope stability issues.

Resource Consent Requirement	**Status**
Capping	
Final capping to be completed within five years of landfill closure.	Time limit extended to 10 years due to capping costs of other landfills in the district. Capping remains incomplete.
Capping to be certified by a registered engineer.	That part of the landfill that has been capped has been certified.
Landscaping will be substantially completed within one year of capping	That part of the landfill that has been capped has not been topsoiled nor landscaped. The majority of the landfill remains uncapped.
Leachate	
Leachate from composting to be drained and either reused or piped to the landfill leachate treatment system.	Leachate from the composting area is currently discharged into the sewer system.
Leachate collection bund to be constructed as per Landfill Management Plan.	No bund has been constructed but a collection drain has been built along the western margin of the landfill.
Stormwater coming in contact with refuse shall be considered as leachate rather than discharged as storm water.	There is currently no control for stormwater runoff.
Leachate collection bund, leachate treatment system and monitoring bores to be installed by 06/04/1996.	Monitoring bores and a treatment system have been installed, yet no bund has been constructed.
Regular water monitoring scheme to be undertaken	See Chapter 5.

TABLE 1.2: SUMMARY OF RELEVANT CLOSURE CONDITIONS FOR THE TAYLOR PASS LANDFILL, BLENHEIM (MODIFIED FROM CONNELL WAGNER, 1998)

Post-closure remedial works

In 1997 the Marlborough District Council commissioned Davidson Partners Ltd of Blenheim, to investigate the then-current status of capping of the Taylor Pass Landfill. The purpose of the investigation was to establish what further work was required to bring the Landfill into compliance with Resource Consent conditions. Discussion and results of investigations are outlined in later chapters.

Partial capping of the southern extremity of the Landfill was completed in 1997. Prior to capping, existing vegetation was stripped and the refuse surface compacted. Capping material obtained from a local subdivision was then spread and compacted in layers to a final minimum thickness of 450mm (Davidson Partners Ltd, 1997a). The capped area remains inadequately vegetated.

The northern end of the Taylor Pass Landfill site hosts a refuse transfer station and composting yard for current waste disposal requirements. Much of the area is asphalt covered. The majority of the area between the composting yard and the capped southern portion of the landfill has been planted in wattle and pine trees.

In 1996, in an effort to reduce leachate emanating from the Taylor Pass Landfill, a basic leachate collection system was installed around the southern and part of the western perimeter of the site. Leachate is collected by means of a series of perforated pipes feeding into a sump/pump system, which pumps the leachate into filter ponds on top of the landfill. Further details of the collection system are discussed in Chapter 3.

1.4.2 Previous landfilling

Prior to filling at the Taylor Pass Landfill site, landfilling occurred beneath the current Brayshaw Park and Riding for the Disabled sites (Figure 1.3 and 1.4). Filling at this northern site commenced in the 1930's, also utilising a disused gravel extraction pit. During the 1940's and into the late 50's, refuse was incinerated on a concrete burning pad, and the ashes were tipped at the entrance to Brayshaw Park (pers com., J.Dovey 1999). No historical records of filling on the site have been located and the extent of filling indicated in Figure 1.3 has been compiled from examination of aerial photographs. Adverse effects on local water from the Brayshaw Park site and the closed Taylor Pass Landfill are difficult to differentiate.

1.4.3 Blue Gums Landfill

Domestic and industrial refuse disposal is now in the Regional Blue Gums sanitary landfill via local transfer stations, one of which is situated on the north end of the Taylor Pass Landfill. The lined Blue Gums Landfill has experienced initial cell leakage problems and although it is also located in the Taylor Valley, it is extremely unlikely that any potential leachate associated with the Blue Gums Landfill will have travelled sufficient distances to affect this investigation.

1.5 Aims and Objectives

The origin and potential impact of the Taylor Pass Landfill leachate on the Blenheim groundwater resource is the focus of this project. The primary objectives are as follows:

- to develop a site hydrogeological model by means of engineering geological mapping, trench logging, geophysical methods, logging of new wells, and correlation of existing bore hole data;

- to characterise the various site cover materials in terms of relevant hydraulic parameters, including the area already capped at the Taylor Pass Landfill site;

- to determine the extent of leachate generation at the Taylor Pass Landfill from shallow groundwater sources derived principally from the Taylor River Fan to enable comparison with direct precipitation-generated sources;

- to determine the nature, extent and rate of migration of the leachate plume formed down gradient from the main Taylor Pass Landfill, including the establishment of a monitoring system; and

- to assess the influence (if any) of the Taylor Pass Landfill and the older landfill site (Brayshaw Park) on groundwater quality in the down-gradient areas, including possible hydraulic connection with the regional aquifer systems.

1.6 Thesis layout

Principles of leachate production and migration are presented in Chapter 2 as an introduction to the problems and issues related to waste disposal. Waste decomposition processes and effects of waste type and age are examined, and water balance modelling is introduced. Chapter 3 outlines the regional geology before more closely investigating local geology and site characteristics. Hydrology and hydrogeology in Chapter 4 sets a basis for assessing characteristic local and regional groundwater flows. The landfill water budget is established, thus predicting leachate production rates, the origin of leachate and potential for contaminant migration. Leachate monitoring regimes and water chemistry test results are discussed in Chapter 5. Geophysical methods used primarily for leachate plume delineation are also briefly discussed in Chapter 5, and results are integrated with hydrogeological and hydrogeochemical results in an attempt to clarify results are integrated with hydrogeological and hydrogeochemical results in an attempt to clarify the extent of the existing contaminant plume. An overview of investigations and project conclusions are presented in Chapter 6.

Chapter 2

Landfills and Leachate

2.1 Introduction

Landfills are the most common form of solid waste disposal and represent a point source of pollution to the environment (Fetter, 1994; Moore, 1990, in Smith, 1992). Post-placement decomposition of waste produces solid, liquid and gaseous by-products from a complex combination of physical, chemical and biological processes (McBean *et.al.*, 1995). The focus of this project is on the effects of the liquid by-product, leachate.

Leachate is defined as a liquid with a high concentration of dissolved solids and/or suspended materials produced by a combination of the percolation of surface and/or groundwater through solid waste, and the decomposition and leaching of solid waste in a landfill (Fenn *et al.* 1975; Fetter, 1994; Smith, 1992; Royds Consulting, 1994). The rate of production of leachate is dependent on several factors: the liquid content, composition and degree of compaction of waste at the time of placement, the volume and rate of precipitation and groundwater infiltrating through the landfill, the presence of materials acting as flow barriers, and temperature. Leachate migration is controlled by the physical and hydraulic properties of the medium through which it is travelling, and attenuation by the physical, chemical and biological environment through which it passes. Leachate composition is a function of both the type of waste present and the degree of decomposition.

Chapter 2 introduces basic concepts of landfill design and operation practices as a background and basis for investigation of site-specific conditions at the Taylor Pass Landfill in the following chapters.

2.2 Waste Characterisation

2.2.1 Waste Terminology and Classification

The term ***waste*** invokes a broad range of definitions. The New Zealand Chemical Industry Council (1991) broadly defines waste as:

> *"unavoidable materials for which there is currently or no near-future economic demand and for which treatment and/or disposal may be required".*

Hazardous wastes is further defined as:

> *"unwanted materials which exhibit hazardous characteristics such as corrosivity, explosiveness, reactivity, flammability or radioactivity, or otherwise have potential to damage human, animal and other species"* (CAE, 1992).

Hazardous materials used in everyday household activities are found in wastes of domestic origin, and thus although a landfill may not be specifically designed to contain hazardous waste, it is inevitable that it will be received. This is especially the case in older landfills where control over filling procedures and protocols have been minimal or non-existent (CAE, 1992).

In order to assess likely effects of waste disposal by landfilling on the environment, it is first necessary to identify the character of waste disposed over time. Where in the past waste has been classified by its origin (i.e. municipal, commercial or industrial), a more appropriate classification system when dealing with site management and pollution potential is to classify waste as biodegradable, non-biodegradable, inorganic, inert, water soluble, acidic, etc (CAE, 1992), thus providing more information on the likely resultant products and rate of decomposition processes within the landfill.

In many cases where landfilling was initiated prior to the recent interest in landfilling procedures and hazards, classification of waste was not accurately carried out. This is the case for many older landfills in New Zealand and as such, estimation of volumes and character of waste disposed has been the subject of considerable investigation. Figure 2.1 shows a breakdown of the character of solid wastes within the total waste stream. Willmot (1991) adopts the analysis of Porteous (1989) in Figure 2.1a which indicates significantly less vegetable matter than Figure 2.1b, which illustrates data from surveys of selected New Zealand centres (Manukau, Rotorua and Christchurch). The variations in comparative proportions of waste makeup illustrated in Figure 2.1 reflect the uncertainty associated with predicting waste composition based on areas with differing social and industrial environments.

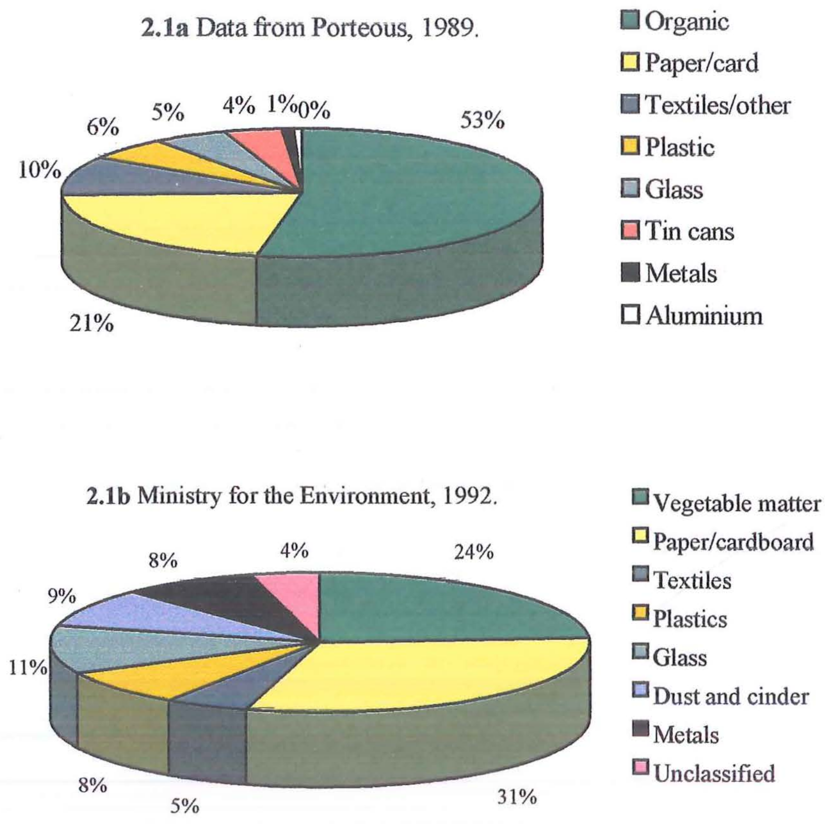


FIGURE 2.1: COMPARISON OF RESULTS OF WASTE STREAM ANALYSES. (FIGURES GIVEN IN PERCENTAGE BY WEIGHT.)

2.2.2 Taylor Pass Landfill

The specific history of filling of the Taylor Pass Landfill is uncertain, but during an assessment of environmental effects for the Resource Consent application in 1994 Royds Consulting noted the following with respect to the acceptance of waste for filling:

> *"The site accepts most waste types, although wastes of a hazardous nature are judged for suitability on an individual basis. Staff have endeavoured to restrict known hazardous waste disposal at the site, although there is an unknown quantity of such wastes buried at the site."*

Taylor Pass Landfill is known to have accepted the following hazardous wastes (Royds Consulting, 1994) but as indicated above, no quantitative data exist:

- waste oil,
- sewage sludge and screenings,
- electroplating waste,
- timber treatment wastes,

- paint waste,
- drycleaning waste,
- asbestos,
- pesticide waste, and
- septic tank waste.

Trenching within the landfill (Appendix 2.1, and further discussed in Section 3.4.3) identified a significant amount of clean fill material, comprised of excavated gravels and soil materials, mixed with general refuse. Once again however, there are no records indicating the percentage of total disposed waste comprising clean fill material. Other non-hazardous wastes can be assumed similar in proportions to the waste stream analyses of the Ministry for the Environment (1992) illustrated in Figure 2.1b, with paper products and organic matter making up a significant percentage by weight. Current activity at the landfill includes disposal of animal wastes and various organic sludges.

The contents of the Taylor Pass Landfill can thus be described as comprising predominantly household and industrial refuse mixed with significant clean fill materials and an unknown quantity of hazardous wastes as listed above.

2.3 Leachate Generation

2.3.1 Mechanisms

A quantitative analysis of leachate volumes is essential for both predicting and understanding the effects of landfilling on water quality, and thus determining the most effective control measures. Leachate generation is a function of refuse conditions, landfill ground surface conditions, underlying subsurface conditions, and the volume of water available to filter through waste and leach contaminants. Water availability is the most critical factor and is largely dependent on the three aforementioned conditions.

Water is supplied for leachate generation by means of precipitation, groundwater infiltration, surface water run-on, water present in refuse on emplacement, co-disposal of sludge wastes, and water formed during microbial decomposition of biodegradable organics (Lu *et al.* 1985; Figure.2.2). The nature of leachate migration within groundwater is dependent on the local hydrogeological conditions, and the density and viscosity of the leachate in comparison to the local groundwater. A leachate plume forms and travels generally in the direction of local groundwater flow. The leachate may separate into a number of phases depending on relative densities and states (e.g. aqueous or non-aqueous). Light non-aqueous phases liquids (LNAPLs) will float at the water surface, whereas dense non-aqueous phase liquids (DNAPLs) will tend to sink through the water column. The resultant leachate plume therefore may form a highly complex stratified body subject to different trends of migration.

Where refuse is disposed of below the local groundwater level, leachate is generated immediately and acts as an instantaneous source of pollution. If emplacement occurs immediately above the water table, groundwater mounding may occur, saturating refuse and also resulting in immediate leachate generation (Lu *et al.* 1985). Leachate generation is delayed where refuse remains above the local groundwater level because infiltrating precipitation and surface flow must saturate the body of waste before leachate-forming liquids are able to percolate vertically under the influence of gravity. The heterogeneous nature of refuse will inevitably lead to some areas of refuse reaching field capacity and producing leachate before others, hence leachate production will not commence simultaneously over the landfill area, nor will leachate composition be laterally consistent.

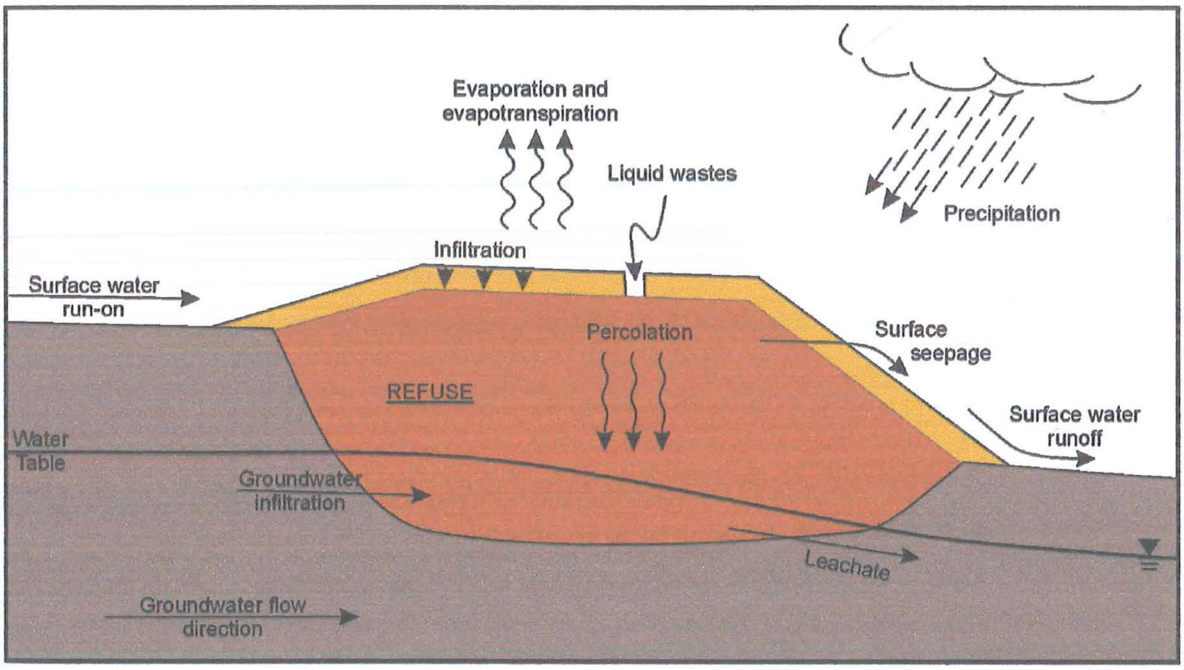


FIGURE 2.2: WATER INFILTRATION AND SEEPAGE AT A LANDFILL (MODIFIED FROM KNOX 1985, AND MCBEAN *ET AL.* 1995).

The volume of water available to infiltrate through a landfill is the primary determinant for the amount of leachate produced. Thus, in a humid environment a landfill is likely to produce significantly more leachate than in an arid environment given the same ground and refuse conditions. Conversely, the concentration of contaminants within a leachate is likely to be significantly greater in an arid climate. Humid and arid climates thus lead to two radically different leachate generation conditions characterised by large volumes of relatively dilute leachate and small volumes of concentrated leachate respectively (Fetter 1994, Fenn 1975).

2.3.2 Leachate Generation Models

A number of methods of water flux calculations have been devised for estimating the amount of water infiltrating through a body of waste to produce leachate. The Water Balance Method (Thornthwaite and Mather, 1957) applied to solid waste disposal sites (Fenn *et al.*, 1975) predicts leachate generation by calculating moisture availability and transfer in soils. The principal source of moisture is precipitation, and thus it is primarily a method for calculating only the surface water component through a single homogeneous cover layer.

The Hydrologic Evaluation of Landfill Performance (HELP) model (Schroeder *et al.*, 1994) allows for modelling precipitation infiltration through multi-layered cover systems (e.g. Figure 2.3), and accounts for both lateral and vertical drainage within cover layers. Berger *et al.* (1996) point out two critical points affecting the efficiency of the HELP method, which are also unaccounted for in the water balance method: a) Gravitational forces only are considered as the driving force for downwards percolation, and b) No consideration is given to the integrity of compacted cohesive clay soil barriers. The water content of the soil has a marked effect in both cases; in unsaturated soils, capillary forces significantly affect water flow, and unsaturated clay soils may form desiccation cracks during dry periods increasing the overall hydraulic conductivity of the cover layer.

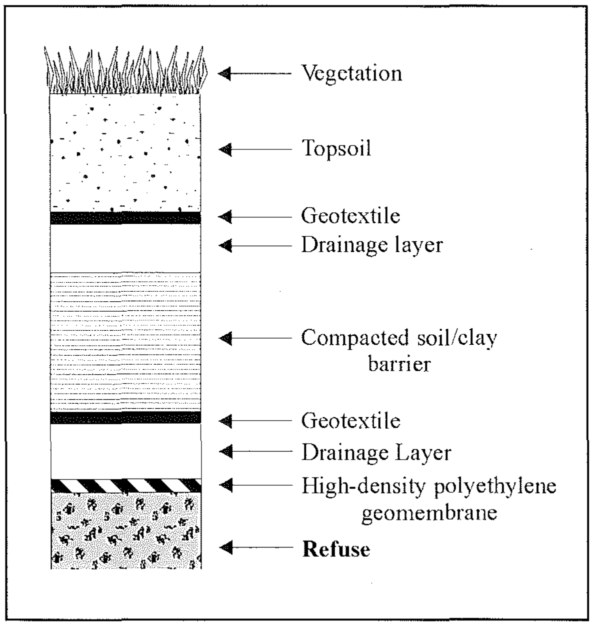


FIGURE 2.3: EXAMPLE OF AN ENGINEERED MULTI-LAYERED LANDFILL COVER INCLUDING BOTH NATURAL AND SYNTHETIC MATERIALS.

The method of analysis for the infiltration through a cover surface should include as much site-specific data as is available, however the complexity and variability of climatic and site conditions leads to uncertainty, inherent to hydrological modelling. Further discussion and calculations by the Water Balance method are presented in Chapter 4.

2.4 Leachate Migration and Attenuation

Migration of leachate through the subsurface is an important concept in trying to identify the location of a leachate plume and to assess the risk posed to surrounding groundwater. Once in contact with the groundwater, leachate will migrate in the direction of groundwater flow away from the source. The presence and nature of the interface between permeable gravels and sands, and relatively impermeable silts and clays, will greatly affect both the lateral and vertical movement of a leachate plume within an aquifer. Thus a reasonable understanding of the three dimensional geology and the nature of groundwater flow must be obtained. These concepts are further discussed in Chapters 3 and 4, respectively.

Natural attenuation occurs as leachate migrates within the saturated and unsaturated materials beneath a landfill. Attenuation is defined by McBean *et al.* (1995) as "the reduction of contaminant concentrations during transport through the soil environment". Thus attenuation is an important factor when investigating the pollution potential from migrating leachate. Biological attenuation mechanisms associated with waste degradation processes are discussed in Section 2.5, whilst the main physical and chemical attenuation mechanisms are discussed below.

2.4.1 Physical Attenuation

Physical attenuation capability is a function of soil particle size distribution and texture, and hydraulic characteristics. Processes include filtration, dilution, hydraulic dispersion, sorption, volatilisation and diffusion (McBean *et.al.*, 1995), some of which are further discussed below.

Filtration

Filtering is the physical removal of suspended solids as the leachate migrates through a medium. The rate of removal of suspended solids from solution is a function of the size of the voids through which the leachate is migrating. Low permeability materials are the most efficient for the removal of colloidal sized particles from suspension (Qasim and Chang, 1994; McBean *et.al.*, 1995).

Dilution and dispersion

Dilution involves the reduction in concentration of contaminants due to physical mixing of leachate and uncontaminated groundwater, the rate of which is proportional to the volumes of both leachate and groundwater. Conversely, dispersion involves the reduction in concentration of contaminants as a result of variations in groundwater flow velocities due to aquifer heterogeneities on a large scale, and to pore channels on a microscopic scale. Although the overall mass of contaminants remains the same for both dilution and dispersion, the concentration of contaminants is reduced by a factor proportional to the amount of groundwater mixing that occurs. For poorly reactive or inert elements and compounds (chlorides, nitrates and sulphates, sodium and potassium), dilution and dispersion are the only means of attenuation (Qasim and Chang, 1994; McBean *et.al.*, 1995; Freeze and Cherry, 1979).

Sorption

Physical sorption involves the attraction of solute to soil particle surfaces by van der Waals forces. Soil particles of organic origin are currently believed to be the primary soil constituents involved in the sorption process (Qasim and Chang, 1994; McBean *et.al.*, 1995).

2.4.2 Chemical Attenuation

A measure of the amount of chemical, in comparison to microbial attenuation can be ascertained by the ratio of COD (chemical oxygen demand) and BOD (biological oxygen demand) in contaminant analysis.

Precipitation

The chemical deposition of solids from solution, termed precipitation, is a significant chemical attenuation mechanism for decreasing contaminant levels in leachate. The pH level of the soil is a major contributing factor, however. Precipitation occurs most often close to the base of a landfill but is inhibited by the accumulation of organic acids early in the life of a landfill. The dissolution of calcium carbonate can lead to neutralisation of the acids, hence promoting precipitation but leading also to the increased "hardness" of the resultant fluid.

McBean *et.al.* (1995) identify two modes of precipitation reaction. The first involves an ion exchange whereby a small ion (e.g. Na^{+}) is replaced by a large ion (e.g. Ca^{+}), causing overall enlargement of colloidal sized particles, and removal from migrating leachate by a filtering mechanism. Hence the process is both chemical and physical. The second precipitation phenomenon is characterised by the formation of insoluble salts of multivalent metallic ions. This in effect means the bonding of a metal cation (e.g. Zn^{2+}, Pb^{2+}, Mn^{2+}) with an anion (e.g. S^{2-}, OH^{-}, CO_3^{2-}) to form a precipitate (e.g. ZnS, $Pb(OH)_2$, and $MnCO_3$ respectively). It has been shown that precipitation is effective in the removal of lead, zinc, mercury, copper and chromium cations from waters (Griffith, 1976; McBean *et al.*, 1995).

Ion Exchange

For any soil, the nature and amount of clay present, organic content and the soil pH determine the potential for a soil to exchange ions. Clay content increases the cation exchange capacity (CEC) both because of the large surface area, and the permanent negative charge held by secondary silicate minerals that make up most clays. Organic content and a high pH also improve CEC (Qasim and Chiang, 1994; McBean *et.al.*, 1995).

Adsorption

Adsorption is the adhering of contaminant molecules to the surface of other particles, effectively decreasing total dissolved solids in a leachate. The mechanism is a function of the surface area of the host soil particles, and hence is most effective in clay soils where the surface area to size ratio is high. Adsorption and ion exchange mechanisms are both pH-dependent processes (Qasim and Chiang, 1994; McBean *et.al.*, 1995).

Diffusion

Molecular diffusion occurs in response to a concentration gradient. Constituents will tend to move from areas of high concentration to areas of low concentration. Diffusion is a slow process, hence a more effective attenuation mechanism when associated with slowly migrating leachate (Qasim and Chiang, 1994; McBean *et.al.*, 1995).

2.5 Solid Waste Decomposition

As solid waste decomposes following disposal, different biologic processes and environmental conditions define four characteristic decomposition phases as follows:

Aerobic Phase.

Anaerobic Phase - Acid Formation.

Anaerobic Degradation – Methane Formation.

Final Maturation.

Decomposition is generally faster under aerobic than anaerobic conditions (Lu *et al.* 1985), and therefore is most rapid immediately upon emplacement of waste. Qasim and Chiang (1994) present generalized degradation curves of a landfill during the decomposition process (Figure 2.4), which illustrates changes in leachate properties characteristic of each decomposition phase.

2.5.1 Aerobic Phase

Initial aerobic decomposition occurs immediately following placement of waste as follows:

$$\mathbf{6\,(CH_2O)X + 5\,O_2 \xrightarrow{\text{(Aerobic microbial activity)}} CH_2O + 5\,CO_2 + 5\,H_2O + 6\,X + ENERGY}$$

where X includes ligands such as PO_4^{3-}, SO_4^{2-}, NO_3^{2-}, NH_3 (Smith, 1992).

Aerobic decomposition is relatively short-lived due to the lack of oxygen within buried refuse, and the high biological oxygen demand associated with this phase. Where refuse is in contact with the atmosphere this phase may continue. Water and carbon dioxide are produced as by-products of this phase of decomposition, however little or no leachate is generated as materials generally remain below critical saturation and moisture will not migrate vertically under the influence of gravity.

Any leachate that is produced during the initial aerobic phase will generally only contain entrained solids and highly soluble compounds, (eg. NaCl). The temperature of the landfill is elevated during this phase as substantial heat energy is produced. Organic by-products are volatile fatty acids, amino acids, carbohydrates and sugars. Oxidisation of metals decreases as available oxygen depletes and the pH decreases below neutral.

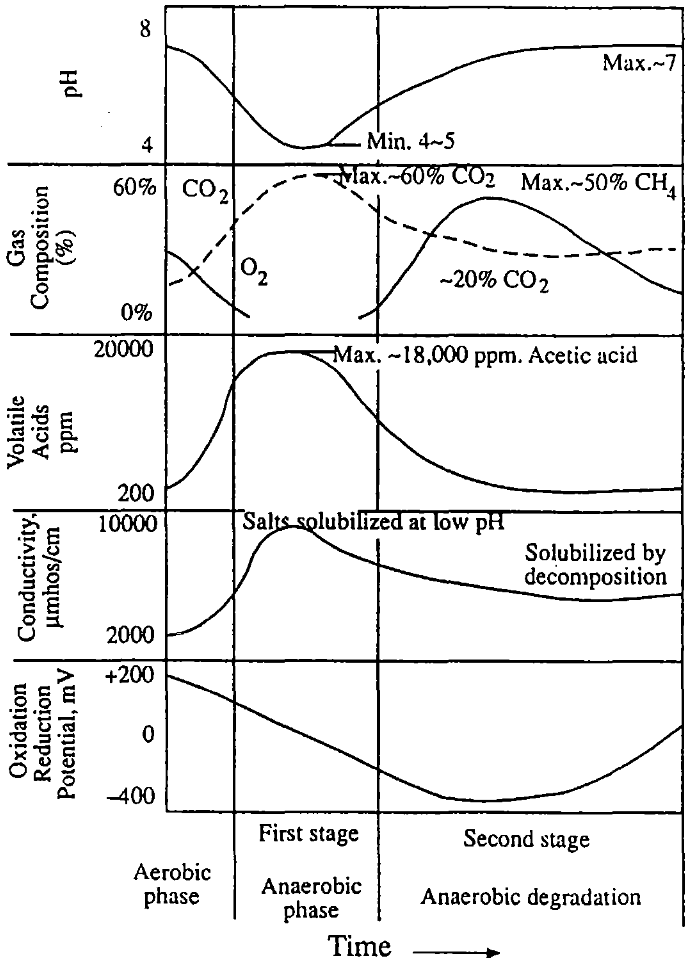


FIGURE 2.4: GENERALISED DEGRADATION CURVES OF A THEORETICAL LANDFILL DURING THE DECOMPOSITION PROCESS. FROM STANFORTH, HAM AND ANDERSON (1979), IN QASIM AND CHIANG (1994).

2.5.2 Anaerobic Phase - Acid Formation

Depletion of oxygen and its replacement by carbon dioxide leads to a decomposition phase involving both anaerobic organisms. Significant amounts of fatty acids are formed (predominantly acetates), carbon dioxide increases and the pH level drops to a minimum, decreasing the sorptive capacity of the refuse and increasing the ion exchange between the leachate and organic matter.

(anaerobes)

$$\underset{\text{Organics in solid waste}}{5\,(CH_2O)X} \rightarrow (CH_2O)X + 2\,\underset{\text{Organic acid}}{CH_3COOH} + 4X + \text{ENERGY}$$

(from Smith, 1992).

Large quantities of inorganics (Cl^-, SO_4^{2-}, Ca^{2+}, Mg^{2+}, Na^+) become more soluble at low pH producing a high ionic strength leachate reflected by high conductivity. Chemical oxygen demand is high. The redox potential is progressively reduced to below zero, which in turn enables the growth of methanogenic bacteria and the onset of an anaerobic degradation phase.

2.5.3 Anaerobic Degradation – Methane Formation

Methane-producing bacteria degrade fatty acids produced in the previous phase by fermentation:

$$\underset{\text{Fatty Acids}}{5\,CH_3COOH} \xrightarrow{\text{(anaerobes)}} 2\,(CH_2O)X + 2\,\underset{\text{Methane}}{CH_4} + 4\,CO_2 + \text{ENERGY}$$

(from Smith, 1992).

Methanobacteria are present only under restricted conditions as follows (Qasim and Chiang 1994, Smith 1992, Lu *et al.* 1985):

- absence of oxygen;
- neutral pH (6.6 to 7.3);
- temperature range 10 to 16°C;
- suitable C:N ratios; and
- sufficient available nutrients.

As methanobacteria establish and progressively utilise organic acids the pH drops, promoting further bacteria establishment and a consequent increase in methane production rates. Reported theoretical methane production rates are highly variable and have been reported from 0.03 m^3/kg to a possible 0.5 m^3/kg waste (Milke 1992, Robinson 1986 in Smith 1992). Methane production may be an issue for anywhere from 10 to 100 years after refuse emplacement.

Carbon dioxide production is reduced during the methane fermentation phase. Near-neutral pH reduces solubilisation of inorganic materials resulting in decreased conductivity also. Surplus CO_2 after hydrogenation to form methane in this phase may be retained as bicarbonate (Baedecker and Back 1979, in Smith 1992). Denitrification causes the breakdown of amino acids, increasing the ammoniacal-nitrogen content of the leachate produced.

2.5.4 Final Maturation

As stabilisation continues, the landfill becomes depleted in nutrients and the rate of bacterial decomposition declines. Aerobic conditions may be partially re-established as meteoric and groundwaters continue to infiltrate the landfill, which becomes more susceptible to infiltration and degradation continues (Qasim and Chiang 1994). Heavy metals may be mobilised as microbes producing humic-like substances attack resistant organics (Smith 1992). The redox potential increases as methane production drops and the landfill becomes stabilised.

The decomposition and degradation process will continue as long as organic matter is available and may take several years to decades to reach completion depending on temperature, water movement, pH, and age, compaction and composition of the solid waste.

2.6 Leachate Characterisation

Four classes of major components in landfill leachate have been identified by CAE (1992) as follows:

1. major elements and ions (e.g. Ca, Mg, Fe, Na, NH_4, CO_3, SO_4, Cl),
2. trace metals (e.g. Mn, Cr, Ni, Pb, Cd),
3. organic compounds often measured as Total Organic Carbon (TOC) or Chemical Oxygen Demand (COD) and some individual organic species (e.g. phenol), and
4. microbiological components (e.g. bacteria, viruses, fungi and parasites).

The variety and range of concentrations of leachate constituents above is highly variable and is influenced by the type and age of waste, its decomposition state, the rate of application of water, refuse moisture, landfill design and operation, and the interaction between the leachate and the environment.

2.6.1 Landfill Operation and Refuse Composition

Operational procedures (e.g. refuse cover frequency and thickness) refuse processing and compaction and refuse compaction directly influence both the rate of water infiltration and the availability of oxygen to decomposing refuse. The depth of a landfill influences leachate composition by influencing the availability of oxygen to microbes. Contaminants released during aerobic and anaerobic decomposition are significantly different. Anaerobic conditions are enhanced in thicker refuse bodies

Comparatively speaking, deep landfills also allow extended contact time between percolating water and refuse resulting in substantially greater concentrations of contaminants than shallow landfills under similar conditions (Qasim and Burchinal 1970, in Qasim and Chiang, 1992). Greater depths also increase the time required for saturation to occur, hence waste stabilisation and contaminant release also occur over a longer period of time associated with percolation of waters through the waste body.

Variation in the municipal waste stream composition was discussed in section 2.3. Similarly, the composition of leachate will vary based on the variability of municipal waste (Bagchi, 1990). The inclusion of hazardous wastes especially will degrade the quality of leached waters. Roger and Totton (1988) suggest, however, that leachates from New Zealand landfills appear relatively insensitive to refuse composition within typical wastestream variations.

2.6.2 Age of landfill

The age of a landfill is a convenient way of expressing its state of decomposition, as microbial activity is the governing factor for the extraction of pollutants from solid waste. Lu *et al.* (1985) carried out an extensive data review of more than 30 sites over a 25-year period. Results indicated that the concentration of organic indicators (biochemical oxygen demand, BOD; chemical oxygen demand, COD; and total organic carbon, TOC) within leachate gradually decline after reaching maximum concentration approximately 2-3 years after filling. Other organic and inorganic constituents decrease in concentration after 3-5 years of flushing of the refuse bed by infiltrating waters. The concentration of heavy metals often fluctuates over the landfill stabilisation period as microenvironments within the landfill mass promote differing conditions of dissolution, precipitation, adsoption and complexation mechanisms (Qasim and Chiang, 1992). These results are illustrated in Table 2.1, which shows typical composition of landfill leachate at different stages of landfill stabilisation. The time-frame over which the decrease in pollutant concentrations occurs varies considerably, with factors including waste characteristics, composition, processing and compaction, and interactions between waste, geological and atmospheric systems. Bacteria survival decreases with the age of the landfill, as growth and survival are inhibited by the high temperatures and low pH produced during early decomposition phases (Ware 1980 in Lu *et al.* 1985). Viruses cannot multiply outside of a host organism and as such are generally absent in landfill leachates of any age.

2.6.3 Rate of water infiltration

Infiltrating water directly influences both the quantity and quality of leachate produced within a landfill. Once field capacity has been reached, the rate of leachate production closely follows the rate of water infiltration, with leachates containing a high contaminant load resulting from the concentration effects of low water infiltration rates. Biodegradation processes, and thus landfill stabilisation, occur more readily where the moisture content of refuse is above 75% of the dry refuse weight. When the moisture content remains above this threshold then constituents are released more rapidly, yet are immediately diluted by high flow rates.

Investigations of pollutant concentration fluctuations associated with seasonal water flux and flow rates have been contradictory, however Akkeson and Nilsson (1997) report a seasonal alternation between methanogenic and acidogenic phases of leachate production based on pH and BOD and COD levels in two test cells containing 4 000 t of refuse each. The different leaching capacities and resulting pollutants associated with the different degradation phases can thus produce seasonally varying leachate.

Parameter	Age of Landfill		
	1 Year	5 Year	16 Year
BOD	7 500 - 28 000	4 000	80
COD	10 000 - 40 000	8 000	400
pH	5.2 - 6.4	6.3	
TDS (Total dissolved solids)	10 000 - 14 000	6 794	1 200
Specific Conductance	600 - 9 000		
Alkalinity (as $CaCO_3$)	800 - 4 000	5 810	2 250
Hardness (as $CaCO_3$)	3 500 - 5 000	2 200	540
Total phosphorus	25 - 35	12	8
Ammonia-N	56 - 482		
Nitrate	0.2 - 0.8	0.5	1.6
Calcium	900 - 1 700	308	109
Chloride	600 - 800	1 330	70
Sodium	450 - 500	810	34
Potassium	295 - 310	610	39
Sulphate	400 - 650	2	2
Manganese	75 - 125	0.06	0.06
Magnesium	160 - 250	450	90
Iron	210 - 325	6.3	0.6
Zinc	10 - 30.	0.4	0.1
Copper	-	<0.5	<0.5
Cadmium	-	<0.05	<0.05
Lead	-	0.5	1.0

N.B. All values are mg/l except specific conductance as microhms per centimetre and pH as pH units.

TABLE 2.1: COMPARISON OF RECENT AND OLDER LANDFILL LEACHATE (AFTER CHIANG AND DEWALLE, IN QASIM AND CHIANG, 1992).

Although variations in leachate quality occur with the rate of water application, the cumulative mass of leached contaminants per unit mass of refuse remains the same for low and high infiltration models. Thus, Lu *et.al.* (1985) suggest that initial high moisture application rates, and consequent rapid removal of contaminants from the landfill body, can reduce the long term pollution potential, provided that short term leaching problems can be controlled.

2.7 Landfill Design and Leachate Control

2.7.1 General Background

A recent rapid increase in published literature on landfill design and practice reflects an increase in environmental awareness and consequent improvement of operation protocols. Landfill design techniques have rapidly developed with the aim to minimise leachate generation and therefore minimise the potential for contamination of soils and water by refuse derived leachates. The formation of some leachate is inevitable as decomposition progresses, and the careful collection and treatment of leachate may become necessary. The following section briefly outlines landfill design and basic remedial techniques. For further discussion the reader is referred to such publications as those by Bagchi (1990), Qasim and Chiang (1992), the German Geotechnical Society (1993), and the Geological Society London (1996), amongst the vast amount of available literature currently available on the operation and construction of landfills.

Construction of a well-designed landfill aims at reducing leachate contamination by minimising the potential for leachate generation and migration. A ***sanitary landfill*** is defined as a disposal facility, which includes the following design control and operation features:

- site lining of either a natural, engineered or synthetic nature;
- leachate and gas control systems;
- comprehensive operational plans including the disposal of waste in a controlled and methodical fashion, and regular monitoring of environmental effects that extends past the closure of the site; and
- closure strategy including final site cover and landscaping to ensure minimum exposure and maximum stability.

Many landfills include a combination of these features, and increasingly landfills are being constructed to implement all of the above criteria.

One of the single most influential design features impacting on the migration of leachate into underlying soils and eventually groundwaters is the nature of the landfill base. Two main types of landfill can be defined based on basal characteristics and their leachate control properties. These are either contained sites or uncontained natural attenuation landfill sites.

2.7.2 Containment Landfill Sites

Containment sites are constructed with engineered soil and/or synthetic liner systems to ideally inhibit the movement of leachate from the landfill site (Figure 2.5a). The design and construction of the liner system is dependant on the site climatological conditions and the extent to which leachate is to be contained. It is unrealistic to assume that a landfill will remain fully contained with respect to either precipitation, surface or groundwater entering the site or leachate migrating away from the site, and the design life of a liner system will be influenced by the anticipated quantity and quality of leachate produced.

Drainage systems installed at the base of the landfill (Figure 2.5a) collect leachate, minimising the volume of raw leachate build up at the base of the contained landfill body. Removal and on-site or off-site treatment of leachate is the most common approach taken for the remediation of landfill-contaminated groundwater (Parker, 1992), and is frequently used to reduce the toxicity of raw leachates.

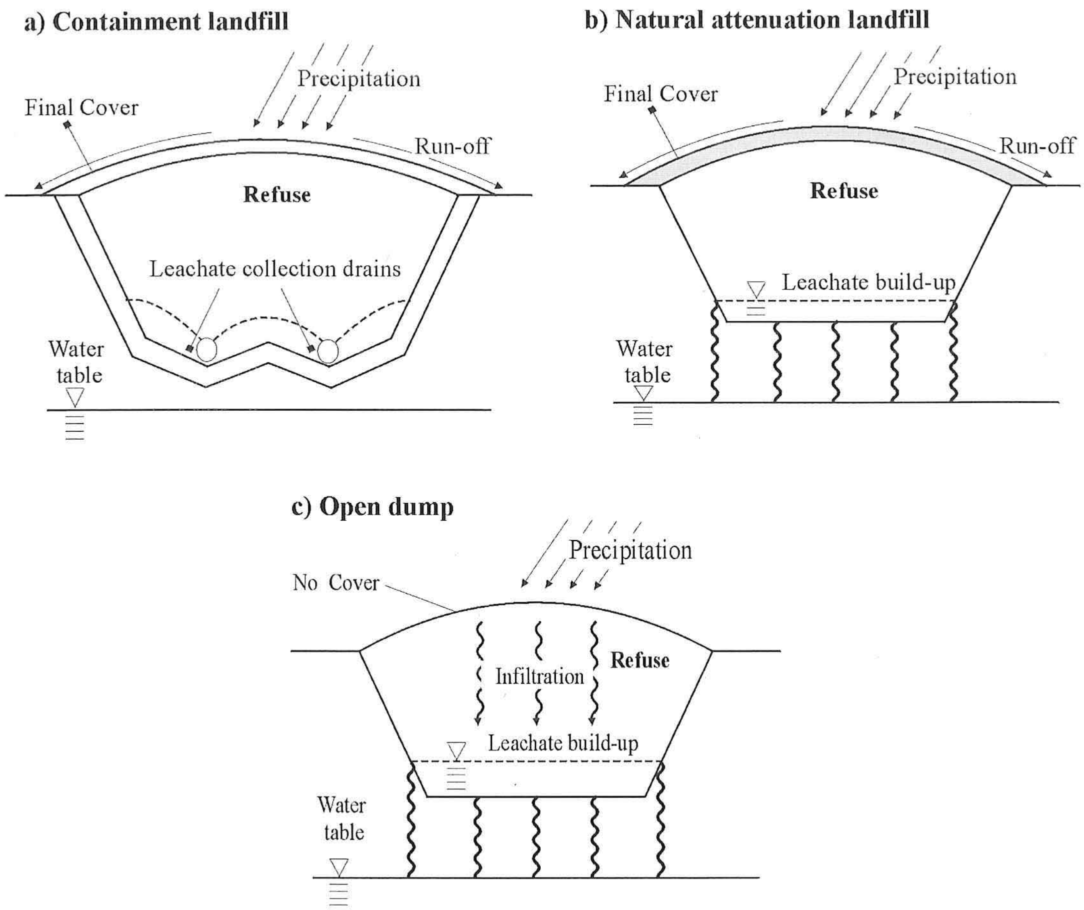


FIGURE 2.5: BASIC LANDFILL DESIGNS FOR DIFFERENT LEACHATE CONTROL PRACTICES.

2.7.3 Natural Attenuation Landfills

Naturally attenuating landfills rely entirely on the attenuation capacities of the natural underlying soils and materials to reduce the contaminant content of leachate to acceptable levels. They contain no drainage system. Leachate formed during decomposition and stabilisation processes is allowed to flow freely from the site at the rate at which it is produced. In order to reduce leachate production rates, a low permeability cover placed over the entire site after closure reduces infiltration of surface waters, hence reducing leachate generation. Waters from decomposition and infiltrating groundwaters remain uncontrolled.

The characteristics of the subsurface geology, water table and groundwater flow beneath a landfill, and anticipated leachate generation volumes, are of great importance in assessing the safety of an undrained landfill. Ideally, the landfills should be located on low permeability material with significant clays for attenuation purposes. The water table should lie below the base of the landfill so that significant attenuation can occur in the unsaturated zone prior to contact with the flowing groundwater. Where the water table intercepts the refuse body, leachate will form immediately following waste disposal and enter groundwater without passing through, and attenuating in the unsaturated zone

Once in contact with groundwater, further dilution occurs a leachate is mixed with uncontaminated groundwater. Monitoring of the leachate plume is necessary to assess the effectiveness of the attenuation system and to determine if further treatment is required.

Open dumps

Common in both urban and rural New Zealand until the 1990's, open dumps are largely unregulated and uncontrolled waste disposal sites. The type of refuse disposed may range from household to industrial waste, and the manner of disposal is in complete disregard for the potential environmental impact. The sites often occupy convenient low spots or holes including abandoned excavation pits without consideration of groundwater issues. Little or no care is given for covering of materials in these situations, hence encouraging vermin and resulting in inevitable odour problems. Infiltration of surface and often groundwater is inevitable, and thus by nature the sites are highly likely to cause contamination of local groundwater at least (Fetter, 1993).

2.7.4 Taylor Pass Landfill

The Taylor Pass Landfill can be classified as a natural attenuation landfill site with only some of the requirements of a sanitary landfill being met. The site was unlined prior to disposal of waste, and originally operated as an open dump with no strict disposal protocol. Measures taken to reduce contamination potential include a partial drainage and leachate recirculation system installed in

1996 and limited capping (Chapter 3), however leachate production and migration remains largely uncontrolled over most of the landfill, as does the groundwater contamination potential (Chapter 4).

2.8 Synthesis

Landfilling ,is the most common form of waste disposal in New Zealand, and liquid leachate produced as a by-product of waste decomposition has the potential to pose a significant threat to the quality of valuable groundwater resources. Many factors influence the production and behaviour of leachate contaminants both within a landfill itself and in the surrounding environment. Thus, a full investigation of surrounding geological and hydrogeological processes and properties, and knowledge of landfill design and operation, is required to fully characterise and understand the extent and potential extent of contamination by landfill leachate.

Waste is defined as those unavoidable materials for which there is no current or pending economic demand, and for which treatment and/or disposal may be required (NZCIC, 1991), with hazardous wastes including those which exhibit objectionable characteristics and may consequently cause harm to people and/or property. The composition of waste disposed to municipal refuse sites is important in assessing the likely composition of leachate produced. However, the composition of leachate appears relatively insensitive to the variation in refuse encountered in municipal sites in New Zealand. It is however important to note that in the case of older municipal landfill sites with minimal control over the acceptance of waste, hazardous waste is likely to be present as a result of general domestic activity.

Generation of leachate from disposed wastes depends predominantly on the amount of water infiltrating through the landfill body, with migration trends dependent on the nature of both the leachate produced and structure of the medium through which it is migrating. A number of models exist for predicting the amount of leachate produced based on water flux calculations. Once a volume of leachate has been produced and begins to migrate both through and away from a landfill, the migrating leachate is subject to a number of processes that alter and attenuate leachate, reducing contaminant concentrations. Filtration, dilution and dispersion are the most important physical attenuation process, whilst precipitation and ion exchange dominate chemical attenuation mechanisms. Biological attenuation occurs as the primary process involved with refuse decomposition.

Solid waste decomposition occurs in four phases controlled primarily by the microenvironment within the body of a landfill, and is distinguished by the composition of resultant leachate contaminants. Initial aerobic-phase decomposition is short lived due to limited oxygen supply and produces minimal leachate containing only entrained solids and highly soluble compounds. Anaerobic decomposition occurs as oxygen is replaced by carbon dioxide within the refuse. The phase is characterised by low pH and the formation of fatty acids. Significant inorganic compounds are mobilised, forming a leachate with high ionic strength and high conductivity. As the pH level approaches neutral anaerobic degradation by methanobacteria degrades fatty acids to produce methane. Leachate produced in this third phase is characterised by high ammoniacal-nitrogen and reducing levels of inorganic contaminants. Final maturation of waste eventuates as nutrients are depleted and bacterial decomposition declines. Moisture infiltration may re-establish aerobic conditions, and heavy metals may be mobilised.

Contaminants within leachate include microbiological components that control the degradation process, major elements and ions, trace metals and organic compounds. The variation in composition and concentration of contaminants is a function of a number of environmental and operational factors. The state of decomposition within a landfill determines the type of contaminants released from a landfill, thus contaminant type can be broadly associated the age of a site. Generally the maximum concentration of organic indicators is reached after 2-3 years with the concentration of other organic and inorganic constituents peaking after 3-5 years. Timing and levels of peak concentration are however highly dependent on the rate of moisture infiltration. Operational practices at and immediately following disposal can have a dramatic effect on the nature of infiltration, and hence on leachate composition. High infiltration rates tend to produce large volumes of low concentration leachate, with low infiltration rates producing small volumes of high concentration leachates. However, the bulk volume of contaminants released over the period of stabilisation remains the same irrelevant of infiltration rates and operational procedures.

Ensuring the safety of the environment is an important concern in modern landfill design. The control of leachate produced by either containment or natural attenuation procedures in sanitary landfill sites is significantly different to the approach taken with older landfills and open dumpsites. The Taylor Pass Landfill includes a partial drainage and recirculation system and limited capping, yet it still relies primarily on natural attenuation mechanisms to reduce leachate concentrations to an acceptable level. Opened and operated as an open dumpsite, the Taylor Pas Landfill has been known to accept hazardous wastes over its lifetime, as the composition of municipal waste disposed to the site was initially largely uncontrolled. The site therefore poses a potential hazard to the safety of local groundwater, and investigations into the geological and hydrogeological environment and chemical characteristics of both the leachate and surrounding groundwaters are required to adequately define the risk associated with any such contamination.

Chapter 3

Geology and Engineering Geology

3.1 Introduction

The American Geological Institute define geology as:

> *"The study of the planet earth—the materials of which it is made, the processes that act on these materials, the products formed..."* (Bates and Jackson, 1984).

Furthermore, engineering geology is defined as:

> *"Application of the geological sciences to engineering practice, to assure that the geologic factors affecting the location, design, and construction of engineering works are recognised and adequately accounted for."* (Bates and Jackson, 1984).

The following chapter then introduces firstly the geological processes and resulting deposits on both a regional and local scale. The changes in the nature of deposits from Taylor Pass Fan to the more dominant Wairau Plains System are investigated primarily by means of well log correlation and geophysical investigation, thus establishing a geological model for the merging of the Taylor Fan and Wairau Plains systems. The geological model later provides a basis for constraint of leachate migration from the Taylor Pass Landfill.

Secondly the Taylor Pass Landfill is examined by means of laboratory based permeability and grain size analysis of cover materials, and on-site mapping and trench excavation to establish the nature of landfill cover and the landfill shape and volume of refuse.

3.2 Regional Tectonic Setting

A series of major transcurrent faults associated with the northern termination of the Alpine Fault controls the geology of the Marlborough Region (Figure 3.1). The Wairau fault trends ENE from the north end of the true Alpine Fault in Tophouse towards the coast at Cloudy Bay. This

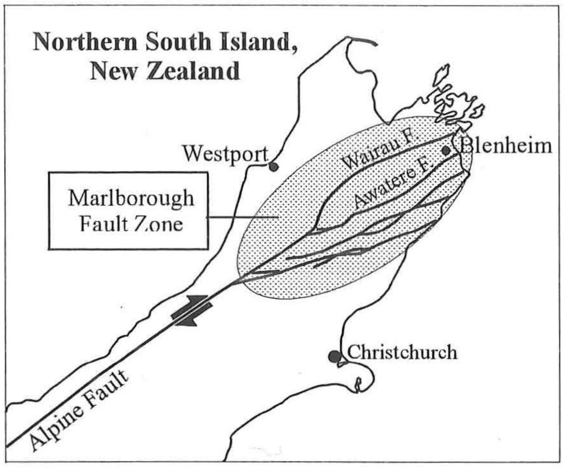


FIGURE 3.1: MAJOR FAULTS OF THE NORTHERN SOUTH ISLAND

northernmost splay of the Alpine fault separates upper Palaeozoic Marlborough schist and greywacke of the Richmond Range in the north from Torlesse greywacke of the Awatere block to the south. Equivalent schists of the Marlborough region located in Southland and Otago indicate progressive lateral displacement of some 480 km along the Alpine Fault.

Differential vertical movement on the Wairau Fault has led to northwards tilting of the Awatere Block bound by the Awatere and Wairau Faults and the creation of an asymmetric valley. Subsequent infilling of the fault angle depression by Quaternary glacial and fluvio-glacial deposits has formed the modern Wairau Plains surface.

3.3 Regional Evolution and Stratigraphy

3.3.1 Basement rocks

Rocks of the Torlesse Supergroup form the basement complex south of the Wairau Fault. Dominated by well-indurated quartz and feldspar rich interbedded sandstones and mudstones, the group also includes conglomerates, limestone lenses, basalt, gabbro and chert (Figure 3.2).

Torlesse rocks are thought to have accreted at the margin of the Gondwana Super continent, derived from an actively rising continental mass. In the upper Wairau, basement Torlesse is late Triassic to Early Jurassic in age, in comparison to the late Jurassic to early Cretaceous lower Wairau rocks which generally contain more plant remains and conglomeratic beds. Significant deformation of the Torlesse Supergroup occurred as a result of continental break-up and the subsequent collision of the Rangitata Orogeny (mid Jurassic to Cretaceous).

The Late Cretaceous and Tertiary Periods saw the onset of accumulation of sediments across New Zealand and in off shore basins, resulting in extensive coal measure and marine sequences. The Kaikoura Orogeny led to erosion of much of the Tertiary coal measure and marine sediments and although still common in many parts of New Zealand, none were preserved in either the Taylor Catchment or the Wairau Region on the whole.

Geological Legend

	Lithology	Period	Era
fa	Unstabilised river silt, sand, and gravel in present river beds.	Holocene	Quaternary
ds	**Dillons Point Formation** — Swamps between beach ridges.		
db	Gravel, coarse sand, and sand of beach ridges and swales and Boulder Bank.		
dl	Sand, silt, and mud lagoon deposits.		
dd	Sand dunes.		
rg	**Rapaura Formation** — Gravel, sand and silt in river flood channels occupied during historic flooding or in modern time prior to flood protection constructions.		
ra_2	Alluvial silt adjacent to present or historic river flood of man-made diversion channels and reclaimed or drained swamps.		
ra_1	Alluvial gravel, sand and silt.		
ra	Alluvial gravel, sand and silt at higher level and well drained.		
sp	**Speargrass Formation** Fluvial (glacial outwash) brown and blue gravel, sand,, and silt. (Otira [last] Glaciation.)	Upper Pleistocene	
to	**Tophouse Formation** Glacial outwash of weathered to slightly weathered brown gravel, sand, and silt; covered by a layer of loess up to 1m thick. (Waimea Glaciation.)		
ma	**Manuka Formation** Glacial outwash of brown-yellow gravel, sand, and silt; covered by 2 to 3m of loess. (Waimaunga Glaciation.)		
wc	**Wairau Conglomerate** Freshwater conglomerate with boulders to pebbles in coarse sand, cemented to semi-cemented matrix covered by loess up to 5m thick.	Pliocene/ Miocene	Tertiary
to	**Torlesse Supergroup** Well indurated, graded bedded greywacke and argillite with basic igneous dykes and extrusives, "cannonball" concretions and rare conglomerate and limestone bands.	Jurassic/ Upper Triassic	Mesozioc
ms	**Marlborough Schist**	Age of emplacement uncertain	Paleozoic

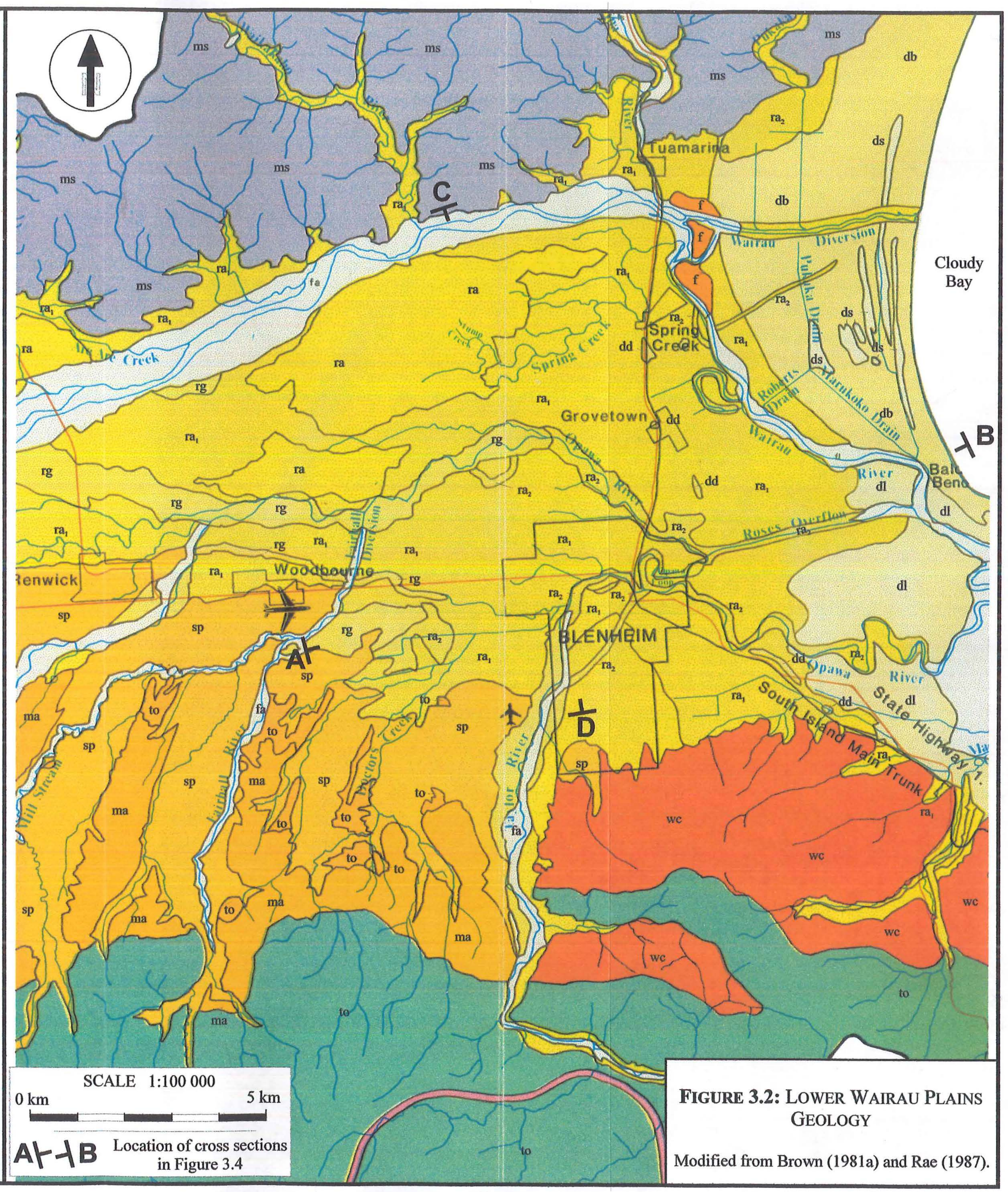


FIGURE 3.2: LOWER WAIRAU PLAINS GEOLOGY

Modified from Brown (1981a) and Rae (1987).

3.3.2 Wairau Conglomerate

Wide spread deposition of Pliocene conglomerates occurred in basins in around the New Zealand region following activity of the Kaikoura Orogeny and erosion of Tertiary sediments. Rae and Tozer (1990) briefly describe the nature and origin of the largely neglected Pliocene conglomerates in the Wairau Region, loosely termed the "Wairau Conglomerates". Predominantly terrestrial in origin, the conglomerates grade laterally into marine conglomerates and siltstones towards the southeast. Late Pliocene deposition occurred in a northwards direction as uplift on the southern side of the Alpine-Wairau fault occurred faster than on the north. Erosion of Torlesse and early Pliocene conglomerates thus formed a series of fans capping the early Wairau Conglomerates. Rae and Tozer (1990) state that the total thickness of the unit varies "from a few to several hundred metres thick" and overlain by "undifferentiated Wairau Gravels".

The early Wairau Conglomerate lies immediately upon Torlesse rocks in the southeast of the Taylor Valley. Outcrops in the Wither Hills region are slightly indurated sandy gravels with weathered to unweathered, subrounded Torlesse clasts up to 20 cm. Secondary calcite commonly cements the coarse sandy matrix. The unit is commonly loess-covered with the thickness of loess being a function of the slope angle, vegetation cover and state of erosion.

Stage	Formation	Duration (years BP)
Aranuian Postglacial – Marine – Fluvial	Dillons Point Rapaura	Present to 14 000
Otira Glaciation	Speargrass	14 000 to 70 000
Kaihinu Interglacial	Winterholme	70 000 to 120 000
Waimea Glaciation	Tophouse	120 000 to 200 000
Karoro Interglacial	Parikawa	200 000 to 250 000
Waimaunga Glaciation	Manuka	250 000 +
Scandinavia Interglacial		
Nemona Glacial		
Unnamed Interglacial		
Porika Glaciation		About 2 000 000 years

TABLE 3.1: GLACIAL AND INTERGLACIAL EVENTS AND DEPOSITS OF THE WAIRAU VALLEY (FROM RAE, 1987).

3.3.3 Quaternary Evolution

Infilling of the Wairau Valley to its current extent occurred as a result of Quaternary glacial and interglacial periods of deposition and associated erosion. Table 3.1 shows the timing of glacial events. As glaciation occurs, vegetation is depleted and considerable sediment is eroded and carried down gradient by advancing glaciers and associated outwash watercourses, and deposited at gradients relative to the corresponding low sea level stand. Conversely, sea level rise and accompanying marine transgression deposition sequences typify interglacial periods, when shortened river courses with decreased sediment load due to decreased erosion, have more energy to down cut into glacial surfaces. The understanding of this cycle and identification of separate units relating to the sequence of events in the Wairau Valley has led to progressive evolution of a geological model of the Wairau Plains (Branch and Dagger, 1934; Wellman, 1955; Suggate, 1965; Browne, 1981a, b; Rae, 1987; Marlborough District Council, 1998).

3.3.4 Glacial deposits

Branch and Dagger (1934) first described the Wairau Gravels and incorporated within the Formation all areas east of the Omaka Valley and raised above Plains level. No differentiation was attempted of the Plains gravels themselves. Subsequent work by Wellman (1955) recognised both the Wairau Surface "...and a higher glacial surface corresponding to the Tophouse Formation". Suggate (1965) was the first to convincingly correlate three glacial surfaces in the Wairau Valley with the recognised Speargrass, Tophouse and Manuka Formations of the neighbouring Buller region. Most recently, Brown (1981b) mapped loess covered surficial gravel, sand and silt deposits of Speargrass, Tophouse and Manuka Formations in the lower Wairau Plains region. Surficial deposits are largely identified by their relative elevations, preservation of the aggradation surface, and thickness of loess cover. Characteristics and occurrences of glacial deposits in the Lower Wairau Valley are listed in Table 3.2.

3.3.5 Post-Glacial Deposits

Dominating the surface geology of the Lower Wairau Plains, modern fluvial, marine and marginal marine deposits of the Rapaura and Dillons Point Formations (Figure 3.2) represent sediments deposited following the retreat of glaciers at the head of the Wairau Valley, following the end of the Otiran Glaciation (*circa.* 14,000 years BP).

Rapaura Formation (Brown 1981a and b)

Postglacial Rapaura Formation consists of up to 30 m of fluvial gravel, sand, silt and clay derived predominantly from the reworking of Speargrass Formation deposits. Revegetation of the upper catchments following the Otira Glaciation (*circa.* 14,000 BP) depleted sediment supply, causing down cutting and degradation of older glacial surfaces throughout the Wairau and tributary valleys, and aggradation towards the coast and in tributary fans respectively.

	Lithology	Characteristics	Occurrence	Age (years before present)
Speargrass Formation	Poorly sorted fluvioglacial gravel, sand, silt and clay. Gravels are fresh blue-grey greywacke.	Depositional surface with a gradient of approximately 1:300 remains well preserved with no loess cover. The Formation is less permeable and more resistant to drilling than overlying Rapaura Formation (discussed in Section 3.2.5), which has a gentler gradient of 1:400.	Forms the main Wairau Plains surface west of Renwick and south of Renwick to the Taylor River. Buried beneath modern Rapaura Formation deposits eastwards of Renwick sloping (1:300) downwards towards the Cloudy Bay Coast where the depth the upper surface is approximately 50-60 m below the ground surface.	14 000 - 70 000
Tophouse Formation	Poorly sorted fluvioglacial gravel, sand, silt and clay. Gravels are predominantly weathered to slightly weathered brown greywacke; schist clasts are commonly restricted to the northern side of the Wairau Valley.	Slightly dissected depositional surface with up to 1 m thick loess cover.	Occurs as inliers surrounded by Speargrass Formation gravels and remnants on the end of Manuka Formation spurs on the southern margin of the Wairau Plains.	120 000 - 200 000
Manuka Formation	Poorly sorted fluvioglacial gravel, sand, silt and clay. Gravels are weathered brown-yellow greywacke; schist clasts are commonly restricted to the northern side of the Wairau Valley.	Highly dissected depositional surface with loess cover up to 2-3 m thick	Located at elevations above 60m, covering the southern hills west of the Taylor River..	250 000 - 310 000

TABLE 3.2: UPPER PLEISTOCENE DEPOSITS OF THE LOWER WAIRAU PLAINS (DATA FROM SUGGATE, 1965).

The Rapaura Formation is divided into two units deposited before (lower) and after (upper) the maximum postglacial sea level (*circa.* 7000 years BP). The lower unit, deposited during the marine transgression and consequent inland degradation/coastal aggradation phase (Figure 3.3a), extends from at least the present coast inland to the Renwick area. The upper unit deposited in a predominantly aggradation and coastal progradational phase (Figure 3.3c) extends inland almost reaching the Waihopai Wairau confluence. Distinguished on the basis of depth, permeability and ease of drilling, the units are difficult to recognise from percussion-drilled bores. Both units comprise generally poorly sorted rounded to subangular gravels to cobble size, in a sand matrix with minor clay and silt. Materials are almost exclusively Torlesse derived.

Overbank deposits form an overlying layer of silts over much of the present surface forming the swamps present over much of the Blenheim area at the time of European settlement and original land surveying. Brown (1981b) describes a wood sample found at a depth of 10.5 m in a bore east of Rapaura, in Late Rapaura gravels dated at <200 years which indicates the rapidity of aggradation of deposits over the Lower Wairau Plains area

Dillons Point Formation (Brown 1981a and b)

Defined by Brown (1981b), Dillons Point Formation is contemporaneous with and separates the lower and upper units of the Rapaura Formation east of Blenheim. The Formation consists of up to 60 m of marine, estuarine, lagoonal and eolian sediments deposited during the postglacial sea level rise and transgression across the Lower Wairau Plains (Figures 3.3 and 3.4).

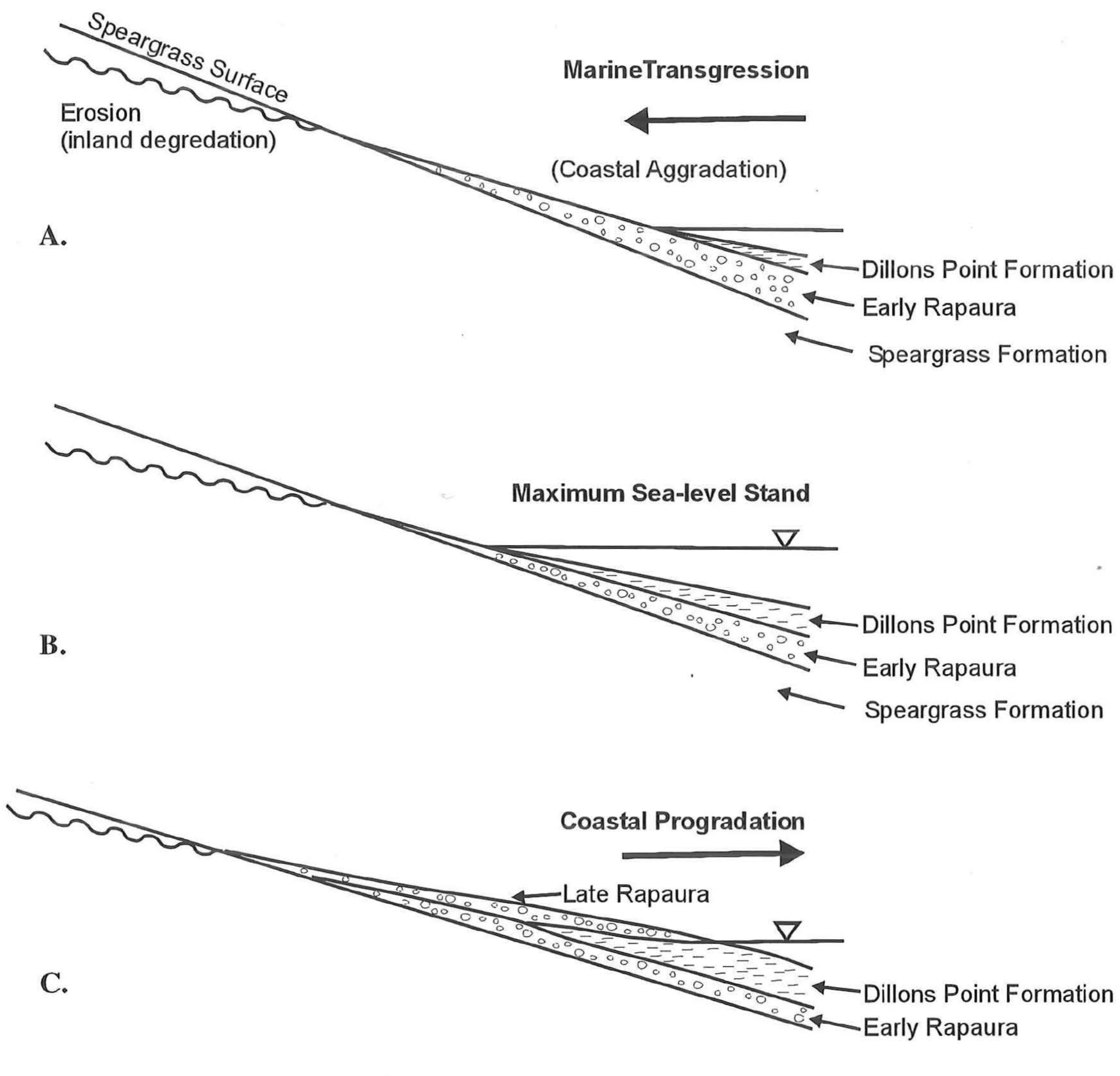
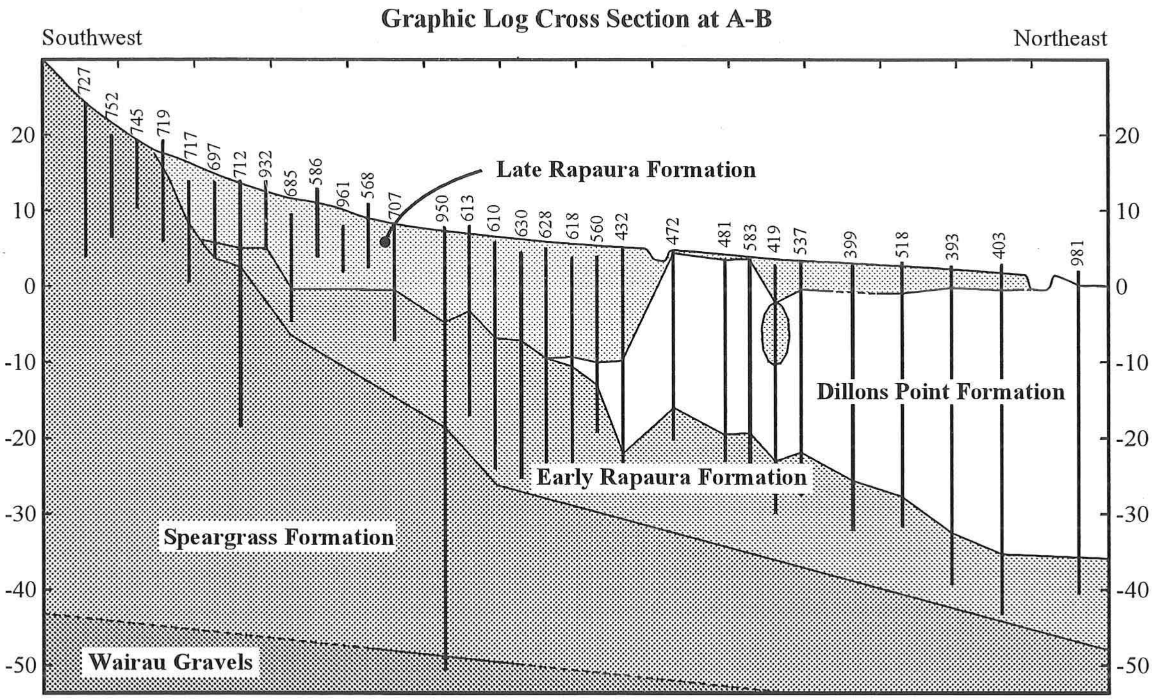
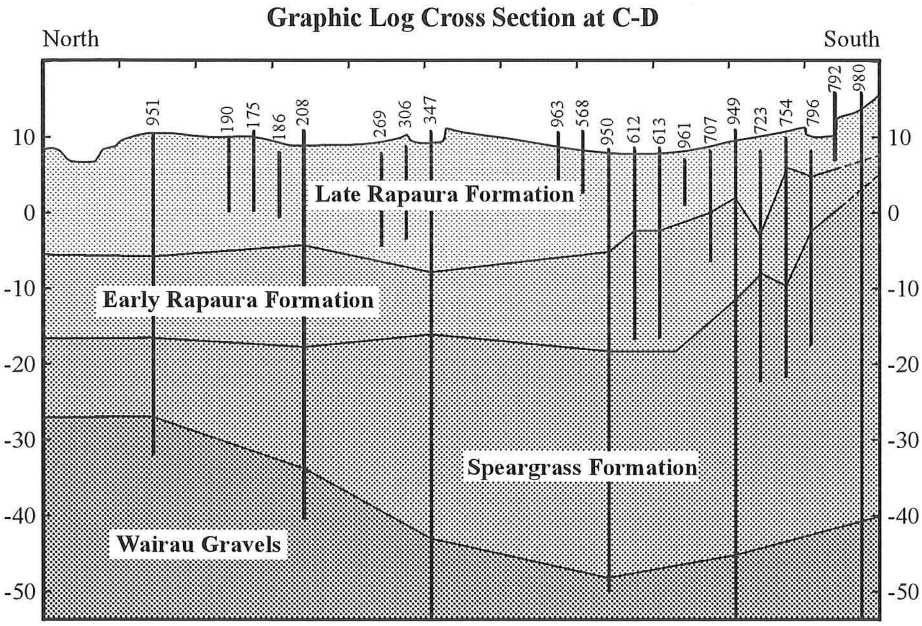


FIGURE 3.3: MARINE REGRESSION AND TRANSGRESSION DEPOSITS ON THE LOWER WAIRAU PLAINS.





Horizontal Scale 1:100 000 Vertical exaggeration 1:100

FIGURE 3.4: WAIRAU PLAINS CROSS-SECTIONS (FROM RAE, 1987). REFER TO FIGURE 3.2 FOR SECTION LOCATION.

The interpreted maximum inland extent of marine transgression is continuously being modified with the drilling of new wells in the district. Brown (1981b) identifies the subsurface boundary as running approximately through Spring Creek to Blenheim, west of the outcropping Dillons Point sand dune band. The westward extent of the Dillons Point Formation interfingers with the Rapaura Formation and is partially dissected by flood channels, which impinged upon the coastal area and marine depositional sequence.

Dillons Point deposits are predominantly beach gravels and sand in the northern coastal reaches whereas finer sand, silt and clay deposits dominate in the south due to the formation of a boulder bank by north flowing coastal currents. Fauna within the northern deposits comprise estuarine, lagoonal, ocean, beach and rock dwelling species in comparison to southern environs, which are restricted to estuarine and lagoonal species. Late Rapaura Formation deposits migrating eastwards have progressively covered Dillons Point Formation deposits except for a mid-lower plains sand dune arc, coastal lagoon deposits, and beach ridges and swales (refer Figure 3.2).

Brown (1981a & b) constructed sections updated by Rae (1987) across the Plains identifying subsurface Speargrass and postglacial deposits (Figure 3.4). No attempt was made to further distinguish deeper "Wairau Gravels". Recent subsurface work by the Marlborough District Council (1998) interpreting deep exploratory and production wells within gravels of the Brancott, Omaka and Benmorven areas have tentatively identified a series of deep glacial and interglacial units (see attached Plan 1).

Distinctive clay layers have been interpreted as being indicative of interglacial deposition of transgressive marine sequences, and are provisionally correlated with events back to the Scandinavian interglacial at approximately 150m below ground surface. The interpretations of MDC (1998) provide a starting point for the correlation of deep wells over the Wairau region. Although the ages assigned by micro- and macro-palaeontological analysis in Plan 1 are reliable, it must be stressed that the correlation of deeper stratigraphic units with known glacial and interglacial stage names is provided only as a *basis* for further analysis. The interpretation and correlation of deep units is continually being refined over time as further data comes to hand (pers. com. Davidson, 2000).

3.4 Taylor Fan Geology

3.4.1 Methodology

A combination of both new and existing geophysical and well log data have been utilised for the investigation and interpretation of the Taylor Fan geology. The aim of the investigation was to determine the type and continuity of deposits associated with the Taylor Fan, the nature of the interface between the Taylor Fan and main Wairau Plains deposits, and to identify any geological influence on the migration and attenuation of leachate.

A brief introduction of alluvial fan characteristics and discussion of well log and geophysical data acquisition, processing and interpretation follows in the section, culminating in the presentation of a current geological model.

3.4.2 Alluvial Fan Morphology

An alluvial fan can be defined as:

> *"a cone-shaped deposit of coarse stream sediments, sheet flood deposits, and debris flows that forms where a narrow canyon stream suddenly disgorges into a flat valley"* (Prothero and Schwab, 1996).

Sediment structures and textures are a function of the change in the hydraulic potential of the stream/river from a higher gradient valley catchment to the flatter valley floor; a steeper fan slope provides greater hydraulic potential, resulting in an overall larger grain size than an equivalent gentle slope.

Sediments are typically poorly sorted with a wide range of grain sizes and little or no organic material is retained (Boggs, 1995). Maximum clast size and average grain size decrease markedly down-slope. In longitudinal profile then, coarse grained debris flows, debris-flow levee deposits and sieve deposits are dominant at the fan head, giving way to finer stream flood and old channel deposits in the distal fan region (Figure 3.5). Vertical sections may show beds with predominant coarsening upwards, coarsening downwards, or little vertical change in grain size. Overall coarsening upwards or downward characteristics depend on the progradational or retrogradional character of fan deposition. Stratification is poor overall, yet may display trough and planar cross bedding and planar stratification with numerous channel structures associated with continuously migrating river channels. A complex series of debris-flow, debris-flow levee, sieve, stream-flood and channel deposits are generally easily discernible on the upper fan surface and a section across the fan profile then will display a sequence of laterally discontinuous debris-flow, debris-flow levee, sieve, stream-flood and channel deposits (Figure 3.5).

The Taylor Fan is a low angle alluvial fan discharging northwards from the uplifting Awatere Block onto the Wairau Plains. Sediments are derived predominantly from Torlesse bedrock, with some recycling of Wairau Gravels, also comprising Torlesse-derived sediments, occurring from the lower catchment valley walls.

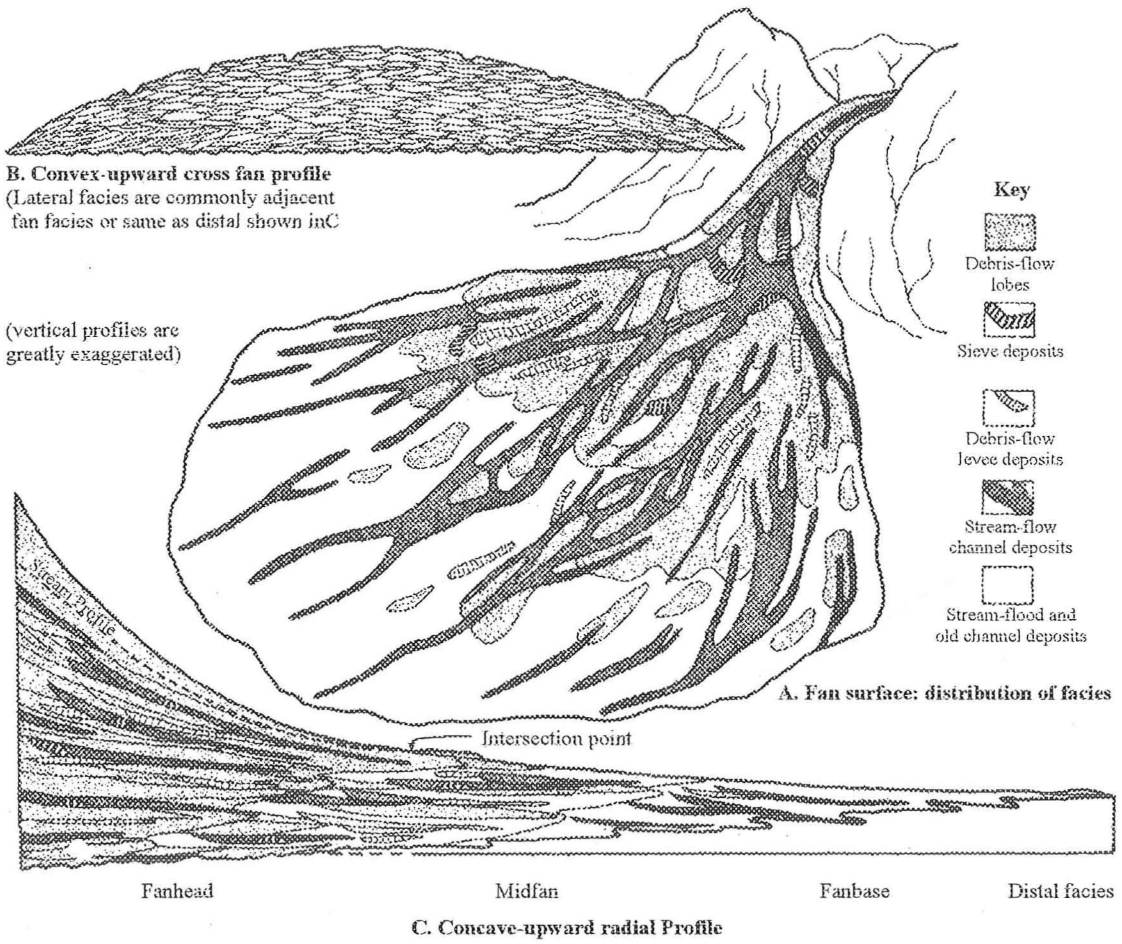


FIGURE 3.5: CHARACTERISTIC ALLUVIAL FAN FEATURES. A) FAN SURFACE. B) PROFILE ACROSS FAN. C) LONGITUDINAL PROFILE. (AFTER SPEARING, D.R., 1974 IN BOGGS, 1995.)

The Lower Taylor Fan Surface geology is dominated by an active Rapaura Formation lobe on the eastern side of the valley adjacent to the Wither Hills. The modern lobe has been forming over 14,000 years, since the end of the last (Otiran) Glaciation. The Taylor Pass Landfill is located within the historic river flood channel of the post-glacial Rapaura lobe. A remnant Speargrass Formation lobe deposited during the Otiran Glaciation (*circa.*14, 000 - 70,000 years BP) flanks the western margin of the Taylor Valley. Uplifted Tertiary Wairau Conglomerate and Upper Pleistocene Manuka and Tophouse Formations blanket the Lower Valley walls (refer Figure 3.2). The Taylor Fan is likely to have been actively aggrading and regarding throughout the Quaternary and it is probable that fan deposits of equivalent age to Manuka and Tophouse Formations (*circa.*250, 000+ and 120, 000 – 200, 000 years BP respectively) exist beneath the Rapaura and Speargrass Formation deposits of the modern fan surface. The depth and extent of any Tophouse and Manuka deposits beneath the Speargrass and Rapaura deposits of the Taylor Fan remains undetermined.

Speargrass Gravels in the Lower Taylor Fan area are poorly sorted, poorly imbricated, fine to coarse gravels in a yellow brown coarse sand to clay matrix. The gravels are rounded to subrounded reflecting both the abrasion typical of a long travel distance from source and the recycling of some sediment. Speargrass degradation scarps remain near vertical due to moderate to tightly packed nature of the deposit and its young age.

On the western margin of the valley floor, Speargrass gravels are mantled by up to 0.5 m of poorly sorted, poorly bedded silty fine to medium sand with rounded to subangular pebbles up to approximately 15 mm. Minor gullies on the western margin of the Taylor Valley have small moderately well vegetated debris/alluvial fans emerging onto the Speargrass surface however no surface water flow is evident and is likely to occur only sporadically. Erosion of loess from the Wither Hills area also contributes fine-grained sediments to the lower fan area, however small tributary fans are not so well preserved due to the absence of the Speargrass surface on the eastern valley margin. Tributary fans from the Wither Hills area are evident along the northern margin of the hills.

The Rapaura surface in the lower fan area is that of a typical braided riverbed with undulating channels and elevated bars trending roughly downstream. The Rapaura gravels are fresh blue-grey to slightly weathered yellow-brown, moderately imbricated, loose, fine to coarse gravels generally in a coarse sand matrix. Fine sediments are absent within the gravel layers, which represent old channel deposits. Fine-grained silts and clays with minor gravel are commonly present as discontinuous sheets and lenses representing flood and overbank sediments deposited by a waning current following a high flow period. During such deposition events the river may have overtopped its cut channel, depositing alluvium over a significant area of the fan and subsequently forming a new river channel.

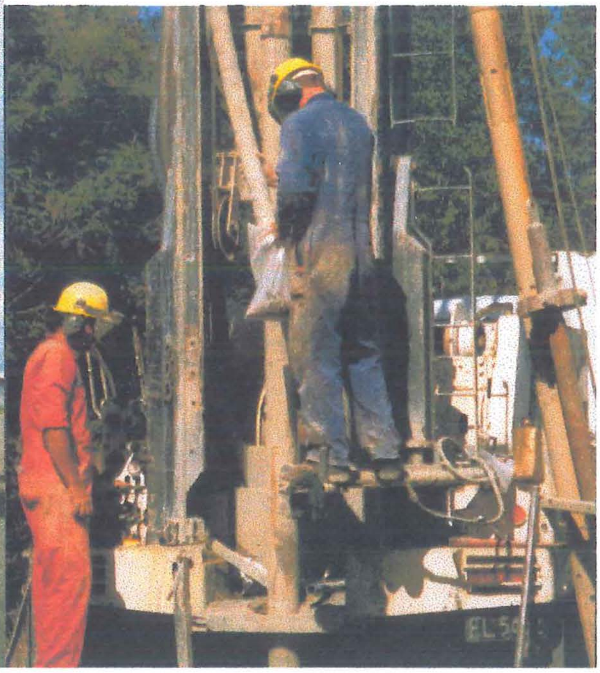
During excavation of the gravel pit in which the Taylor Pass Landfill is now situated, a hard blue impervious clay/silt material ("blue pug") of unknown thickness, but which is thought to represent a fine grained overbank flood deposit, was found covering 100-150 m from the southern end (J Dovey pers.com, 1999). The same deposit has been identified in the Taylor Riverbed adjacent to the southern end of the landfill and at the base of trenches 3 and 4 (Appendix 2) on the eastern side of the landfill. The deposit is thus at least partially laterally continuous for approximately 150 m. Similar deposits have been noted in at number of locations and at various depths in the area (eg. refer well log P28/W0949, Appendix 3), however, no other fine-grained deposit were found to be continuous over the same distance either in the field; lateral continuity of both coarse and fine-grained deposits was seen to be only of the order of metres to tens of metres. The capacity for fine-grained layers to restrict vertical movement of any leachate emanating from the Taylor Pass Landfill then is likely to be minimal

3.4.3 Local Well Data

New Wells

Six wells located primarily for water sampling purposes were installed in the Taylor Fan area using a compressed air driven rotary percussion rig (Figure 3.6). Logging of materials extracted was carried out onsite, with samples being taken at between 1 and 3 metre increments or where a change in lithology was noted. All six wells were constructed to the same basic design; construction details are illustrated in Appendix 3.

FIGURE 3.6: WELL INSTALLATION.
PAGE STREET WELL P28/W3389 (LEFT),
SAMPLE ACQUISITION (BELOW).



Existing Well Data

The Marlborough District Council maintains a data bank of wells drilled in the Wairau Plains region, which was accessed for this research. Due to the lack of demand for groundwater and the comparatively poor yielding nature of the Taylor Fan Aquifer in comparison to the Wairau Aquifer (Chapter 4), deep wells suitable for correlation purposes are sparse on the Taylor Fan Surface. Most wells are of the order of 10 – 20 m deep and located sufficiently far apart that they offer little or no use for correlation purposes within the discontinuous sediments of the Taylor Fan.

Figure 3.7 shows the spatial distribution and depth of wells with obtainable lithological logs. Corresponding section lines are given in attached Well Log Sections. There is a distinct lack of wells in Redwoodtown and Witherlea, and in the area bounded by Ben Morven and New Renwick

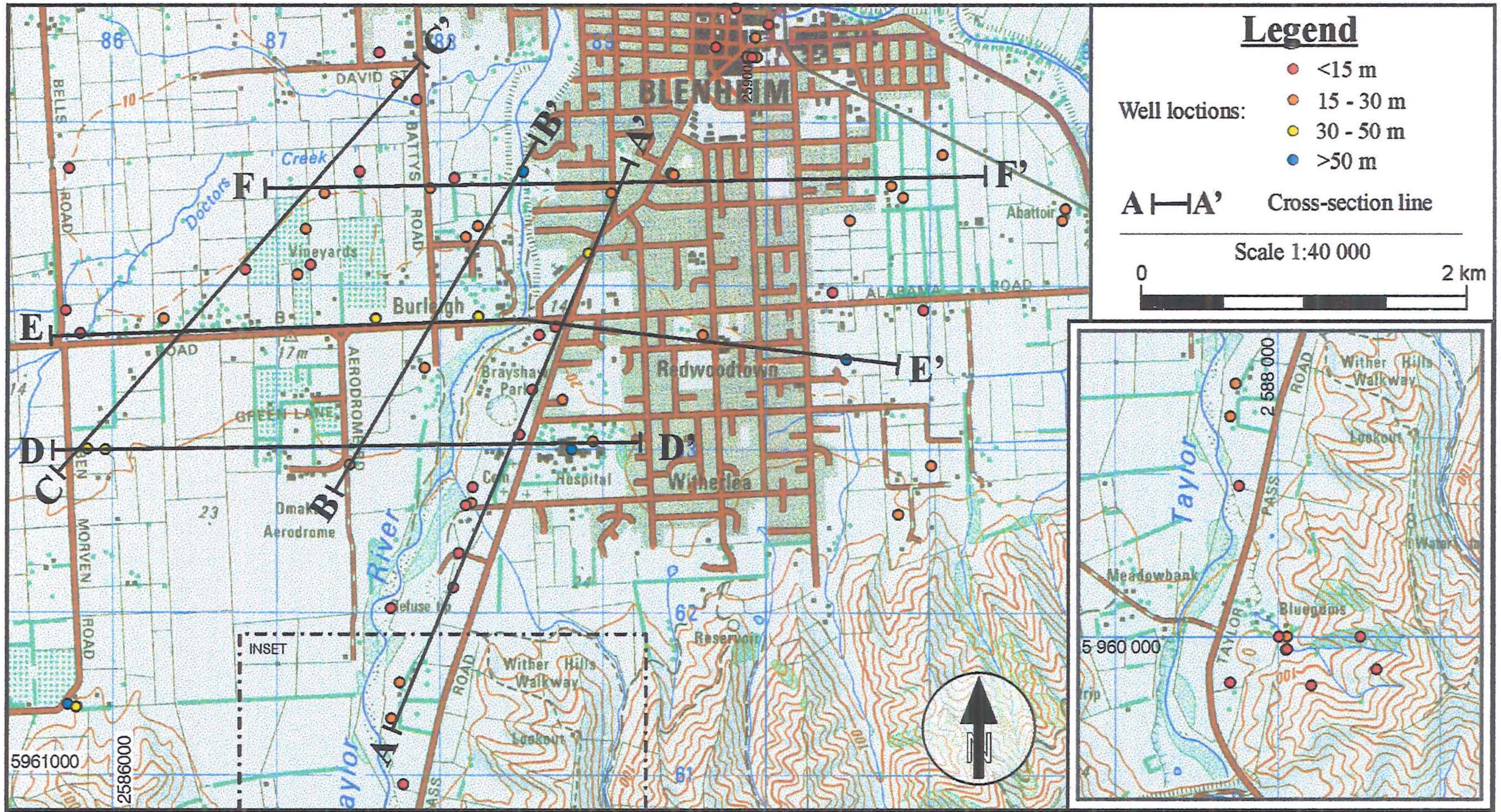


FIGURE 3.7 : SPATIAL DISTRIBUTION AND DEPTH OF WELLS WITH LITHOLOGICAL LOGS

Roads and the Taylor River. The wells forming a NNE trending line between these two areas are predominantly less than 15 m deep and show no noticeable continuity of layers between wells within Taylor Fan deposits (Well Log Section A-A'). The often-apparent "randomness" of sediment texture and discontinuity of layers in the Taylor Fan is representative of fan deposits as shown in Figure 3.5. The distal facies of the Taylor Fan adjacent swamp and fine grained deposits which form a veneer across the Blenheim area are identified in Sections A-A', B-B'.

Cross section F-F' reveals the continuity of the shallow fine-grained deposits across the Blenheim area as previously mentioned in Section 3.2.5. The deposits are interpreted as fine-grained distal fan facies of the Taylor Fan grading into swamp deposits.

Difficulties with correlation of shallow wells in the Rapaura lobe of the Taylor Fan are highlighted in Section A-A'. Whilst constructed cross sections indicate the continuity of some interglacial layers beneath the Taylor Fan surface, it must be emphasised that the correlations are only preliminary correlations based on available data and boundaries indicated in sections are tentative. Deposits indicated as continuous are likely to be is likely to be highly dissected by channel deposits associated with the active alluvial fan system.

3.5 Geophysical Investigations

3.5.1 Background

Geophysical investigations are commonplace in engineering and environmental investigations for a number of reasons:

- they are relatively inexpensive in comparison to other subsurface exploration methods,
- they provide information on a large spatial area whilst remaining largely non-invasive, and
- they are easy to use and offer relatively rapid data acquisition.

The major disadvantage of geophysical methods is in the non-uniqueness of field results. That is, a set of results obtained for any given situation may be interpreted in a number of different ways depending on assumptions made relating to the nature of subsurface materials. Thus any interpretation of geophysical data *must* be correlated and integrated with actual field data in the form of well logs or trenches and the extrapolation of surface geological features in order to provide a sound model that is consistent with all available geological evidence. The use of more than one geophysical method in an area also helps to constrain and verify possible interpretations.

Another disadvantage inherent to geophysical investigations is the hindrance posed by geological and anthropogenic noise. In the case of this investigation, the main sources of noise (and hence the main obstacle to the acquisition of coherent geophysical data) have been the presence of buried and surface power and service cables and electric fences, and considerable amounts of often small yet geophysically significant scrap metal located around the industrial area downstream of the Taylor Pass Landfill and in the Taylor River-bed. The presence of scrap metal in the riverbed is due to the past practice of stockpiling of scrap metal (e.g. car bodies) adjacent to and between the landfill and the Taylor River. Also, the natural movement of gravels has partially buried minor amounts of refuse adjacent to, and likely sourced from, the Brayshaw Park Landfill site. The effect of anthropogenic noise is thus evident at a number of sites due to the predominantly residential/industrial nature of the field area.

3.5.2 Previous Resistivity Investigations

Geophysical work in the Blenheim area has previously been carried out by Groundwater Consultants (NZ) Limited (1983) for the now superseded Marlborough Catchment Board (MCB). A reconnaissance electrical resistivity study was carried out focussing on the New Renwick Road and Battys Road areas in order to assess both the usefulness of the technique for subsurface investigation of groundwater resources without time consuming and expensive drilling programs, and as an assessment of the feasibility of a potential production well located in the Battys Road area. Groundwater Consultants found the method successful in differentiating recent clean gravels of the Rapaura Formation from older silt-bound Speargrass gravels, with resistivity ranges as shown in Table 3.3.

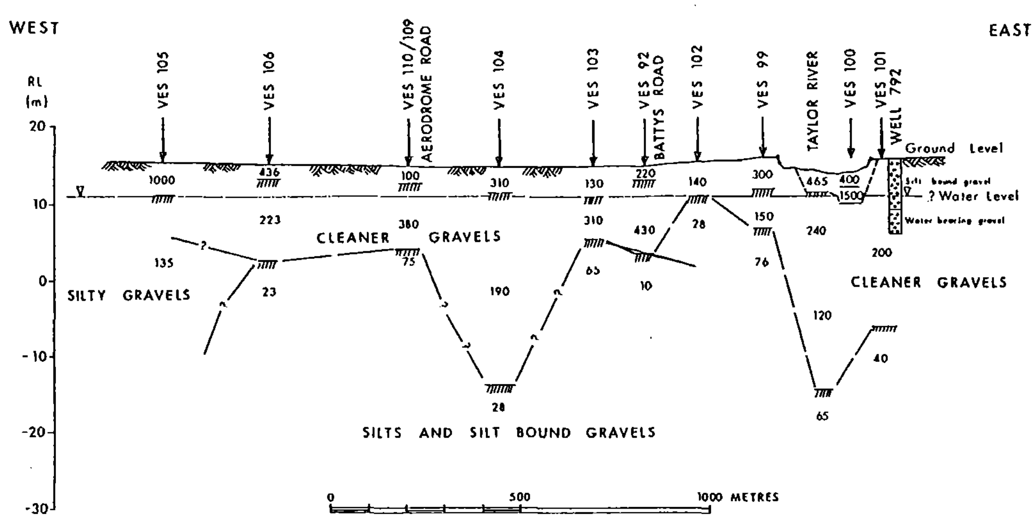
Resistivity (ohm-metres)	**Sediment Type**
150 – 1500	Silt, clay sand and gravel, above the water table.
150 – 430	Silt, sand and gravel of the Rapaura Formation.
30 – 150	Silt, silt-bound gravels and sands of the Wairau/Speargrass Formation.

TABLE 3.3: RESISTIVITY OF SEDIMENTS IN THE LOWER TAYLOR FAN AREA (FROM GROUNDWATER CONSULTANTS LTD, 1983)

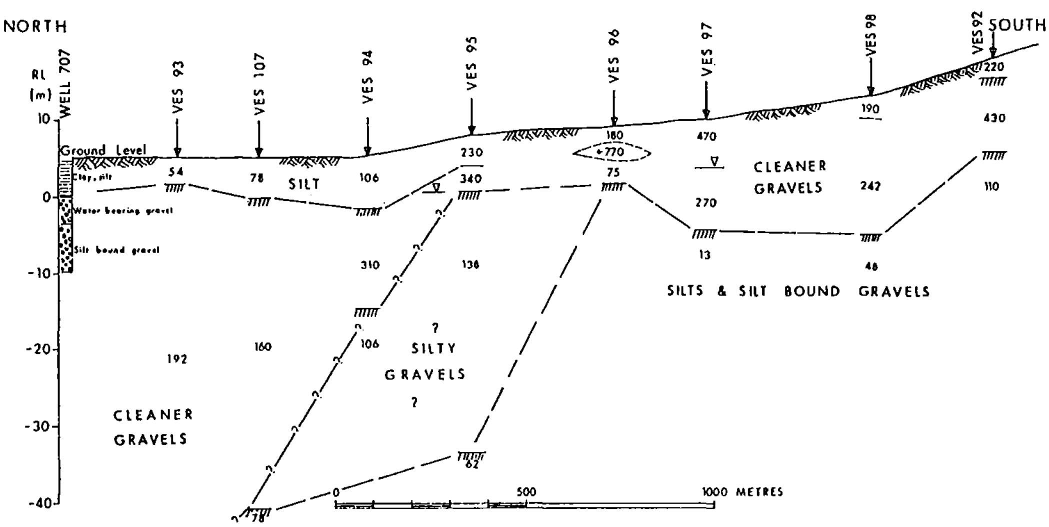
Two perpendicular survey lines were run along Battys Road and New Renwick Road. The New Renwick Road survey line (Figure 3.8a) identifies a thin veneer (5-15 m) of clean Rapaura gravels overlying silt bound Speargrass gravels. Deep sections of Rapaura gravels cut into Speargrass gravels are interpreted as being old channels of the Taylor Fan. Along Battys Road the relatively shallow boundary between clean gravels and silt and silt-bound gravels deepens from 7 m to 46 m over 550 m in a southwards direction. The boundary represents the northern extent of the Taylor

Fan, with the cleaner gravels being indicative of the more active Wairau braided river sequence. The nature of the boundary in Figure 3.8b is however over simplified with the questionable silty gravel deposits more likely to be a gradual transition between Taylor Fan and Wairau deposits represented by a series of interfingering silty and clean gravel lenses.

The investigations carried out and the results obtained by GCL were meant as a reconnaissance study into the feasibility of the resistivity method for groundwater resource evaluation without well drilling, and as such there is little correlative work with well data. The results are therefore not sufficiently constrained by either alternative geophysical methods or subsurface geological data in the form of well logs.



3.8A



3.8B

FIGURE 3.8: LOWER TAYLOR FAN RESISTIVITY PROFILE LINES. A) NEW RENWICK ROAD SECTION; B) BATTYS ROAD SECTION. LAYER RESISTIVITY GIVEN IN OHM-METRES. (FROM GROUNDWATER CONSULTANTS LTD, 1983).

Following the apparent success of the resistivity method for determining stratigraphy in the Taylor Fan/Wairau Plains gravels, further resistivity surveys have been undertaken by the author, and their application to this project are further discussed in the following sections, and in Chapter 6 with reference to plume delineation.

3.5.3 Electromagnetic Methods

Electromagnetic methods (EM and TEM) respond to the electrical properties of the subsurface with responses being dominated firstly by any metals present and secondly by clay content and soil structure, the presence and quality of groundwater and degree of saturation. Hence EM and TEM have proven useful tools for shallow and deep stratigraphic investigations respectively, and the delineation of both leachate plumes and landfill parameters in a number of landfill investigations (eg. Armstrong, 1993; Cardarelli and Bernabini, 1997). A description of EM and TEM theory, and TEM field apparatus and modelling procedures, can be found in Appendix 3. For further discussion on EM apparatus and modelling procedures, the reader is referred to Kearey and Brooks (1991).

An experimental EM31 survey was carried out in April 1999 to establish the effectiveness of the method for identifying channels and other structures capable of acting as a preferential conduit for leachate migration. The initial survey was conducted in a grid pattern across an old surface channel in an area upstream of the landfill where the influence of leachate would not disguise any response from subsurface structures. Both quadrature and in-phase responses proved erratic in the area and showed little or no correlation to each other or the surface channel. Despite trial surveys being carried out in what was thought to be a "noise free" area, responses were in fact generated by shallow buried metal debris common in the area (pers. com. Mark Smith – Taylor Pass Farm Manager). Results were thus incomprehensible and a poor indicator of structure due susceptibility of the method to noise. EM thus proved to be a poor investigative tool for in this area and hence has not used for further investigation.

TEM pilot studies proved effective for defining subsurface geological interfaces when combined with well log data. Both 40 x 40 m and 80 x 80 m transmitter loops were used in order to obtain penetration depths sufficient to target deep (>25 m) boundaries. Although TEM was less susceptible to anthropogenic noise than conventional EM methods, the area required for transmitter set-up greatly restricted the use of the method in the downstream residential and industrial areas. Discussion of specific TEM surveys and results are presented in following sections.

3.5.4 Transient Electromagnetic and Resistivity Surveys

Following previous work by GCL and pilot field investigations, TEM and resistivity were identified as the most useful geophysical tools for the determination of subsurface geology in the Taylor Pass Landfill area. TEM is the primary geophysical method that has been used, with resistivity soundings carried out at a number of TEM sites (Figure 3.9). Results have been correlated with available well log data in an endeavour to determine the geological structure and continuity of layers in the Lower Taylor Fan area.

Survey Layout

Vertical TEM soundings (refer Appendix 3) were carried out in locations identified in Figure 3.9 Where possible, 80 x 80 m transmitter loops were used; where restrictions on accessible area prevented use of the larger loop size, 60 x 60 m or 40 x 40 m loops were used. Figure 3.9 also indicates those TEM sounding locations where resistivity surveys were carried out.

Vertical resistivity soundings were laid out with lines approximately parallel to the down fan direction, as the continuity of deposits in an alluvial fan environment is likely to be greater in the longitudinal direction than in a cross fan direction (refer Figure 3.5). By surveying in the direction of most continuity, resistivity soundings will allow current to travel through more uniform layers, thus producing a truer picture of subsurface structure.

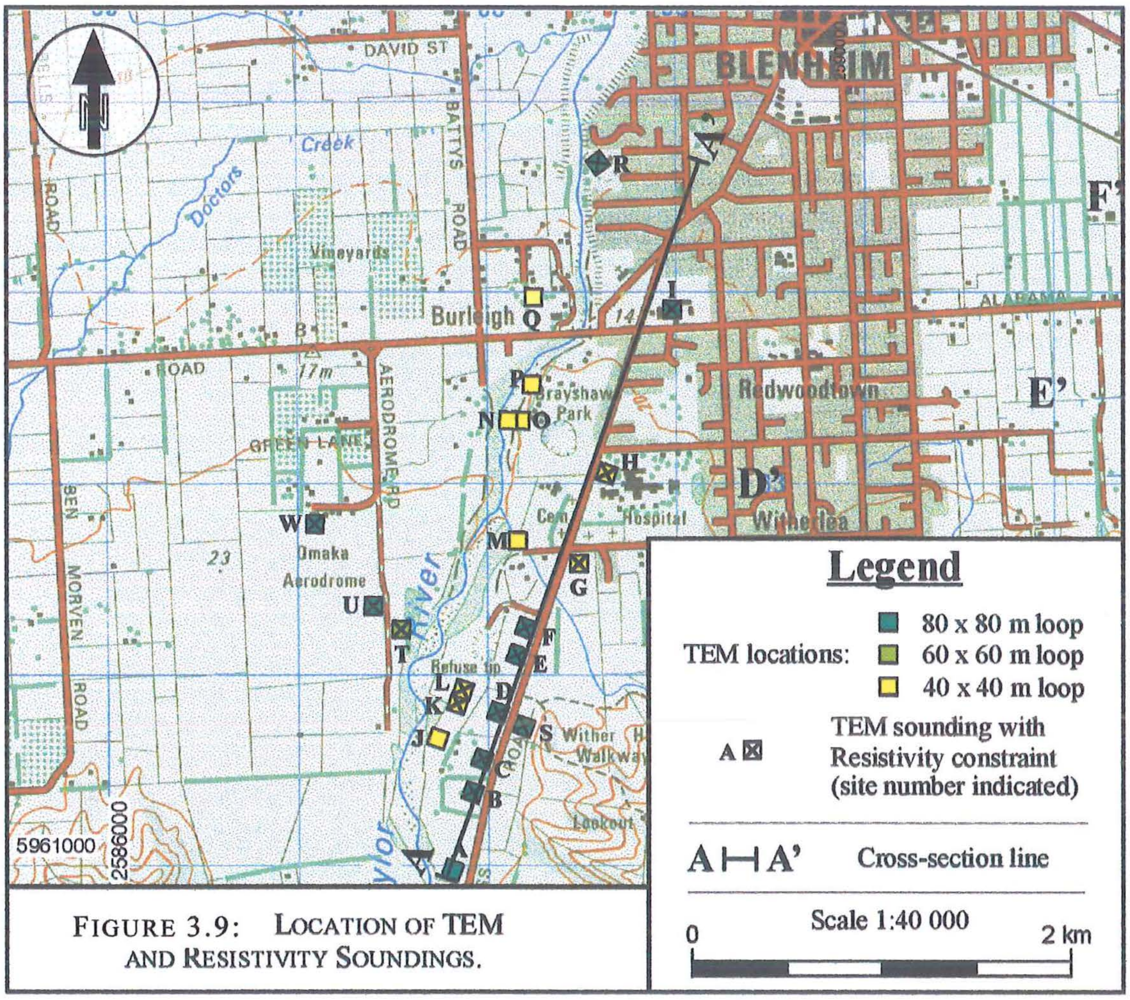


FIGURE 3.9: LOCATION OF TEM AND RESISTIVITY SOUNDINGS.

Data Quality

Both TEM and resistivity data were of a variable standard. Limitations in the practical electrode spacing of between 7 m and 56 m for resistivity surveys allowed a comprehensive depth of penetration of the order of 10 m. Resistivity soundings provided generally smooth data however, with cultural noise affecting only stations G and S (refer Appendix 3). All resistivity data were of sufficient quality to provide essential constraint for the top 10 m of TEM soundings.

TEM data, with deeper signal penetration and larger survey area required per sounding, were also more susceptible to anthropogenic noise. Figure 3.10 shows a comparison of raw data and corresponding best fit and equivalence models for sites B and N, representing good and poor data sets respectively. Site B shows gradually decreasing resistivity with depth indicated by the smooth trend of data from three consecutive time windows (refer Appendix 3). Conversely, Site N displays an unintelligible trend where response from only one time window was obtained, with poor early and late time responses surrounding comprehensive mid-time window readings. Modelling of the Site N data produces a vertical electrical profile that is very difficult to fit within the present geological setting based on resistivity values from other soundings in the area and typical resistivity values.

Other TEM data sets display incomprehensible data at certain times, yet masking of erratic data allows modelling to be carried out with some degree of certainty. Scattering of data points at late times (refer Figure 3.10) indicates a loss of signal at depth; scattered points cannot be modelled and are masked correspondingly. A full set of resistivity and TEM results, are given in Appendix 3.

Constraining Technique

Modelling methods using TemixGL® and ResixPlus® computer software are detailed in Appendix 3. Lithological well logs were used to provide geological constraint for the modelling of resistivity data. Best fit and equivalence model resistivity values of the upper 10 m were then transferred to the upper ten meters of the TEM models. Increasing or decreasing trends in resistivity data below 10 m were also noted and applied to TEM models. TEM data then was modelled using available lithological logs and resistivities provided by resistivity surveys, with resulting resistivity values kept within the typical ranges for clay and alluvium displayed in Figure 3.11.

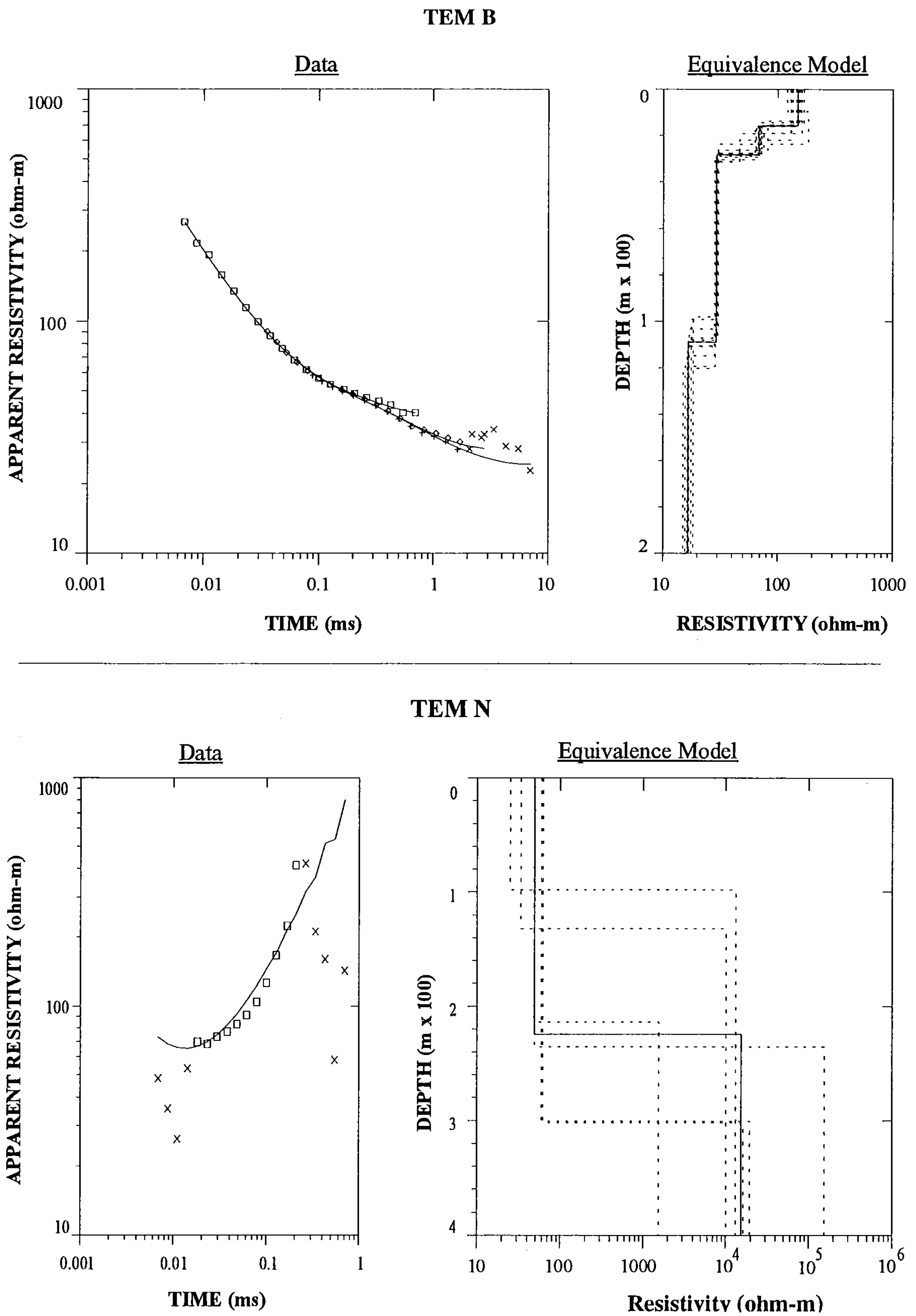


FIGURE 3.10: COMPARISON OF TEM DATA QUALITY AND CONSEQUENT MODELLING OF LAYERS.

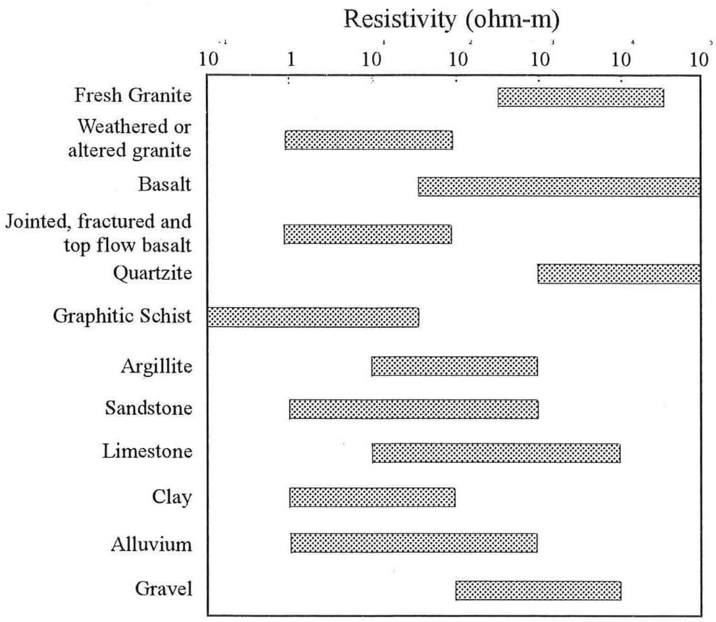


FIGURE 3.11: TYPICAL RANGES OF RESISTIVITIES OF ROCKS AND SOILS (WARD, 1990).

TEM Interpretation

For interpretive purposes TEM soundings located along Taylor Pass Road have been plotted as a "geo-electric cross section" which offer a comprehensive visual correlation technique for extrapolating layers between TEM sites (Figure 3.12). The method allows for immediate inspection of the change in electrical structure with depth and errors in both depth and resistivity of layered models. Construction of further geo-electric cross sections was considered to be unwarranted based on the poor quality of data and limited spatial distribution of sites. Selected TEM profile sites are briefly discussed below and the reader is referred to Appendix 4 for raw data and equivalence models from all sites surveyed and/or discussed below.

Figure 3.12 reveals a steady decrease in resistivity with depth in the Taylor Fan gravels up gradient and in the vicinity of the Taylor Pass Landfill (Sites TEM A, B, C, D). There is no evidence for isolated clay-rich layers, which manifest as highly conductive layers in electrical sections. This is a result of the likely thin nature of any such deposits in an alluvial fan and the inability for TEM to resolve these thin layers at depth. TEM I indicates a conductive layer at 20m which has been tentatively correlated with the Kaihinu Interglacial. Another high conductivity layer at 85 m that has been tentatively correlated with the Karoro Interglacial as located by MDC, 1998 (refer Plan 1).

FIGURE 3.13: LINE A-A' TEM TRANSECT (with incorporated well logs) - Taylor Pass Road

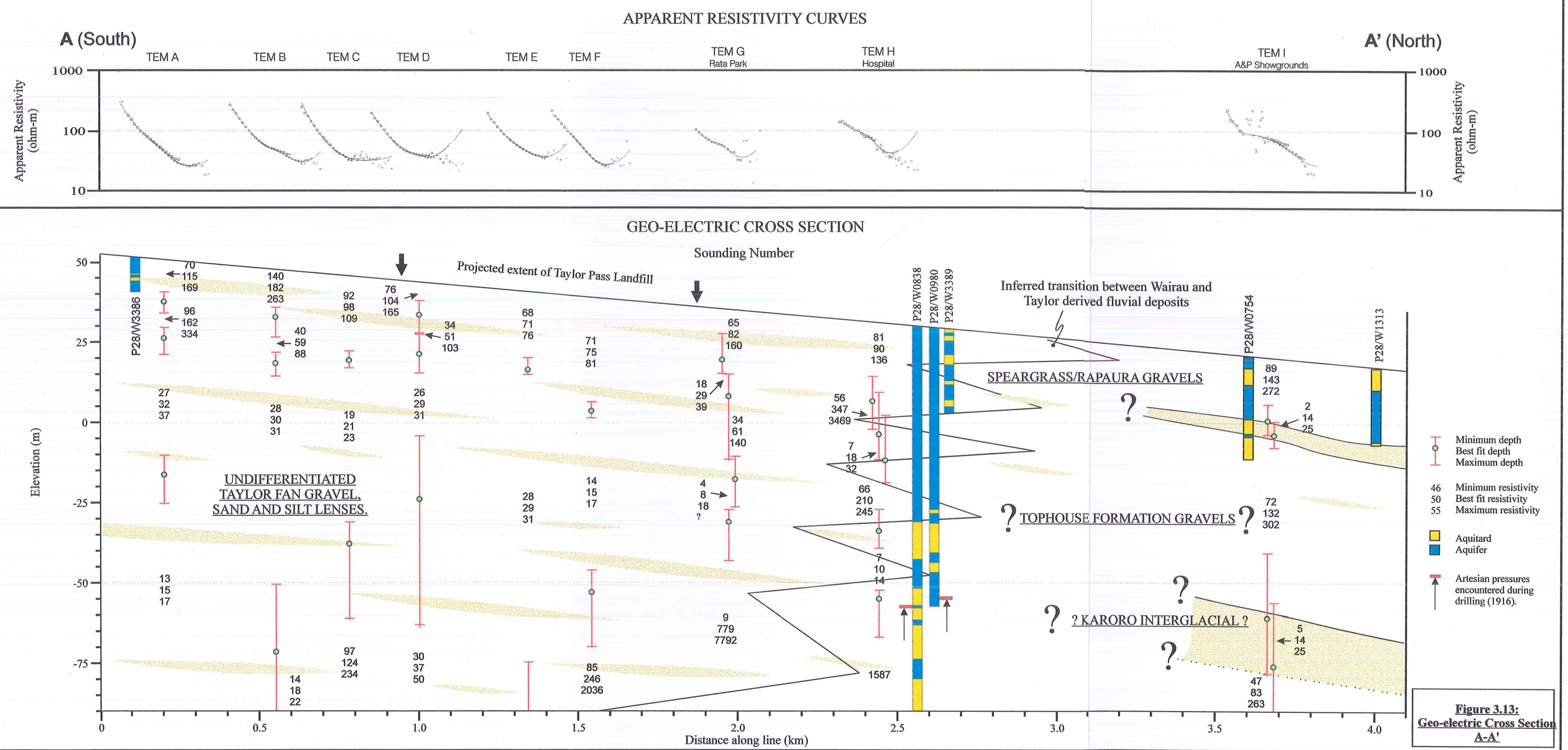


Figure 3.13: Geo-electric Cross Section A-A'

Site TEM A shows an increase in resistivity in the region of 145 m depth. Raw data at this location is comprehensive to the end of the second time window ($t \approx 3$ ms) and displays a definite "kick" in apparent resistivity. This boundary may represent the bedrock alluvium interface in the area however caution must be exercised in this interpretation, as there are no deep wells in the area and thus no constraint to the depth of bedrock. The electrical signal is rapidly lost beyond this depth.

TEM soundings carried out on top of the landfill (TEM K and L, Appendix 3) were aimed at trying to establish the electrical structure beneath the landfill base. However, the highly conductive nature of the landfill itself effectively traps the transmitted signal and prevents any significant response from underlying layers. Layers modelled beneath the landfill base at a depth of 10 m are correspondingly unreliable. To the west of the Taylor River, TEM T, U and W all exhibit a conductive layer of variable thickness between approximately 10 and 30 m. The conductive layer is unconstrained by well log data and as such no convincing correlation can be made. The layer may represent a change in pore water quality rather than a change in lithology. This prospect is further discussed in Chapters 5.

Of significant interest in the Lower Taylor Fan Area is the interface between the Taylor Fan and Wairau Plains gravels. At Site TEM Q, similarities with the electrical structure identified by GCL (1983) are noted in the top 25 m. Although of highly dubious nature due to the effects of noise, data from Sites TEM Q and N both indicate a marked increase in resistivity below 20 – 25 m, which is interpreted as representing clean Wairau Gravels underlying the poorly sorted yet generally finer grained deposits of the distal fan region. The Taylor Fan is therefore inferred to be migrating out over the clean Wairau gravels in a progressive interfingering phase.

A low resistivity layer of the order of 10 to 50 ohm-m at or near the surface is seen at Sites TEM M, O and P may be indicative of poor groundwater quality and is further discussed in Chapter 5. A similar low resistivity layer at TEM R is interpreted as being a continuation of the distal fan facies. A number of wells from New Renwick Road northwards towards Blenheim also reveal shallow fine-grained deposits, sometimes cut and overlain by thin veneers of cleaner gravels. The shallow fine-grained layer extends across the Blenheim Township area to various depths with most recent fine-grained sediments deposited in the swamp environment encountered upon settlement in the Blenheim area (refer Sections 3.3.5, 3.4.3 and Well Log Section A-A').

A relatively sudden drop in resistivity is noted at Sites TEM G (44 m), M (42 m), O (46 m), H (56 m), P (54 m) and R (64 m). This boundary has been tentatively identified as the Karoro Interglacial correlating to that identified by MDC (1998) in Plan 1. The extent of the deposit in the direction of and into the Taylor Fan area is far greater than that of the last interglacial, which is identified only to the New Renwick Road, and further indicates the migration of the Taylor Fan over top of the Wairau Gravels.

3.5.5 Local geological model

Figure 3.13 shows a schematic interpretation of the geological structure of the Lower Taylor Fan area based on and extrapolating between all available well log and geophysical data that have been discussed in the previous sections. In the Taylor Fan itself, deposits are seen to be largely discontinuous with the exception of infrequent flood and overbank deposits such as that represented by the "blue pug" layer underlying the Taylor Pass Landfill. Even the continuity of the fine overbank deposits is likely to be cut by various flood channels.

At the present surface, the distal facies of the Rapaura lobe of the Taylor Fan at New Renwick road grades into fine-grained deposits of the recently swampy margin between the clean Wairau Gravels and the Taylor Fan. The true nature of the interface between the Wairau and Taylor systems is thought to be a complex series of distal fan and river gravel deposits interfingering with each other in response to such factors as sediment supply, rate of uplift and sea level stand.

The continuity of the Karoro (?) Interglacial layer beneath the Taylor Pass Fan suggests progressive migration of the Taylor Fan towards the Wairau Plains since the end of the Karoro Interglacial period at 200,000 years BP. The interaction between the Taylor Fan and main Wairau Plains deposits are further discussed in Chapters 4 and 5 based on hydrogeological and hydrogeochemical investigations respectively.

3.6 Taylor Pass Landfill Site Characteristics

3.6.1 Previous investigations

Davidson Partners Ltd (DPL), Blenheim carried out preliminary geotechnical investigations of capping materials in 1997. Test pits were excavated over the landfill area in order to establish the cover layer thickness and type, and these data are re-evaluated in the following section.

Permeability tests were carried out by DPL (1997) to establish the hydraulic conductivity of landfill cover materials. Falling head permeability testing of three "fully compacted" samples of "light brown silty gravel" yielded permeabilities of 1.34 x 10-8 m/s, 5.05 x 10-9 m/s, and 3.5 x 10-8 m/s, all of which comply with resource consent requirements of 10^{-7} m/s or less. Currently however, the cover material remains only lightly compacted, thus DPL attempted a falling head permeability test on one sample of "lightly compacted" sample however no meaningful results were obtained as the sample remained too permeable to measure by this method. It was predicted that the permeability of tested soils would however be greater than 10^{-7} m/s and further testing by the constant head

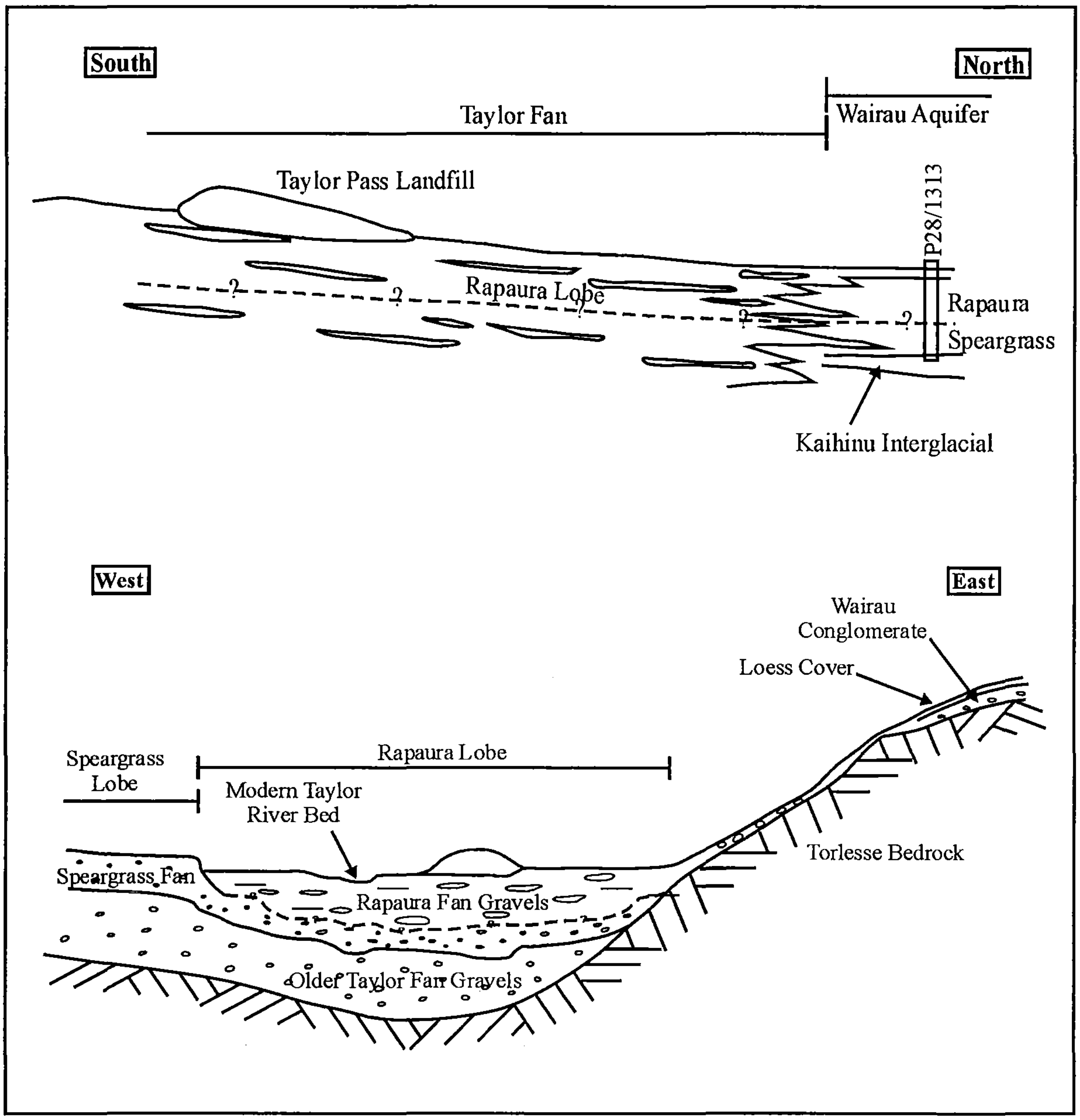


FIGURE 3.13: SCHEMATIC GEOLOGICAL MODEL OF THE LOWER TAYLOR FAN

permeability test method recommended. Further permeability testing carried out by the author and discussion of results are in Section 3.6.2.

Results of the DPL (1997) investigations identify areas of non-compliance relating to both permeability and slope angle. No areas met consent requirements with respect to material permeability, although it was concluded that with sufficient compaction, areas covered with the "light brown silty gravel" would meet permeability criterion.

Side slopes range between 1:2.2 and 1:6 (DPL, 1997; Connell Wagner, 1998) where resource consent requires a maximum grade of 1:3 based on long-term stability and taking into consideration the dry nature of the landfill, moderate slope heights (Connell Wagner, 1998). Top slopes were found to range from less than 1:20 to 1:80. Minimum surface grades of 1:20 are required for rainwater run-off purposes.

Davidson Partners Ltd suggest both the addition of extra cover material where required, along with reshaping and compaction of new and existing cover materials to comply with current consent requirements.

3.6.2 Landfill Cover Materials

Landfill Cover Vegetation and Soils

Mapping of landfill cover materials was undertaken in addition to the re-evaluation of 59 test pits (Appendix 3) excavated and logged by DPL (1997), in order to further establish the suitability of existing cover materials and to identify those areas in which cover materials were obviously substandard. Table 3.4 sets out the 6 mapping units used in Figure 3.14. Due to the relative importance of vegetation on the efficiency of a landfill cover, areas were mapped primarily on the basis of the amount and type of vegetation, and secondarily, the type of soil cover and its susceptibility to erosion. The results of mapping on this basis are further utilised in the assessment of the landfill water budget (Section 4.5).

The poorly vegetated southern portion of the landfill is capped with loess sourced from local residential developments and excavation. The lack of an arable topsoil layer over the top of the loess cap hinders vegetation growth and aggravates erosion of the reconstituted loess soils in the form of minor surface rilling. Rilling depth is currently only of the order of 50 mm however left poorly vegetated, the effectiveness of the cover will continue to deteriorate. The integrity of the cap is also questionable during extreme dry periods when desiccation of the clay rich soils occurs.

FIGURE 3.14: TAYLOR PASS LANDFILL COVER MAP

Mapped units	Description and Occurrence
Exposed refuse	Includes area occupied by offal pits and refuse excavated during the construction of offal pits.
Unsealed tracks	Includes tracks and areas where vegetation growth is inhibited and cover soils are compacted to a certain extent by traffic flow.
Sealed tracks	Includes bitumen sealed entrances to the mid portion of the landfill and access ways through the transfer station. The composting pad is also included as it is constructed with a reputedly impermeable clay lining.
Poorly vegetated soils	Includes areas where vegetation is sparse due to nature of soil cover, or location0 on soil stockpile or near active industrial sites area or
Soils with grass/weed vegetation	Includes a substantial part of the mid and southern landfill areas where off-site earthworks and construction debris including clean-fill and hard-fill have been stockpiled for landfill cover, and landscaped areas surrounding the transfer station.
Soils with trees/shrubs/ and grass vegetation	Includes those area planted with wattle and pine trees through the mid landfill area and along the eastern and northwestern landfill boundaries.

TABLE 3.4: TAYLOR PASS LANDFILL COVER MAPPING UNITS

Surface erosion is not so obvious over the larger area of the landfill due the increased coverage of protective vegetation that no doubt also largely masks the nature of the underlying materials. Vegetated soils over the uncapped area of the landfill are however less likely to be effected by desiccation and cracking.

DPL (1997) excavated and logged 59 test pits over the landfill area (Figure 3.14 and Appendix 3) to determine the nature and depth of cover materials. Test pits reveal that a light brown silty gravel of variable density covers much of the landfill surface (DPL, 1997), with Test Pit 47 at 2.1 m still not penetrating through the cover fill layer. Significant amounts of various fill materials encountered are poorly described in test pit logs, but contain no refuse. Test pits range in depth from 100 – 2100 mm and reveal gravelly clay, buried topsoil, and clay horizons. General refuse is noted in only three test pits (Test Pits 12, 16 and 42); buried decomposed vegetation also occurs in only three test pits (Test Pits 25, 38 and 39); buried wood shavings are noted in Test Pit 7 and surface manure is present in Test Pits 30 and 38. Therefore although the landfill has not technically been capped it is covered to a large extent with variable combinations of gravel, sand, silt and clay (discussed in the next section) to depths of up to and possibly greater than 2.1 m. For the purpose of water balance analysis in Chapter 4 an average depth of cover has been reasonably assumed to be 0.7 m, corresponding to one third of the maximum depth to which refuse-free cover has been identified.

Grain Size Analyses

Soil samples were taken for grain size analysis and permeability testing from eleven sites across the landfill area as indicated in Figure 3.14. Analysed grainsize increments follow those of the British Soil Classification System (BS 5930:1981); analysis methods and results are given in Appendix 3. The samples were all hand excavated from depths of less than 200 mm, and it is assumed that the samples are representative of the "silty gravel" described by DPL (1997).

Results indicate a variable trend in grain-size over the landfill. Samples from within and near the capped section (TPSS-1, 2 and 3) have a high silt fraction (30 - 40%) and variable clay up to 23.7%. Samples TPSS-4 and 6 taken from areas occupied by fill material stockpiles have relatively low gravel content and elevated sand, and silt and clay fractions respectively. In contrast, sample TPSS-5, also taken from within the stockpile area contains 37.5 % gravel, reflecting the heterogeneous nature of the stockpiled material. The irregular surface of the stockpiled material is also likely to be subject to erosion causing removal of the clay silt fraction, with accumulation of finer grained materials in the relative lows of the hummocky surface. Samples TPSS-9, 10 and 11all have a gravel fraction above 50% and corresponding low clay fractions.

Cover soils can therefore be described texturally as sandy clayey silt and gravel in the capped area, and as ranging from clayey silt with gravel and sand (TPSS-6) to sandy gravel with silt and clay (TPS-9, 10 and 11).

Sample	**Grainsize*** (% by weight)			
	Gravel	Sand	Silt	Clay
TPSS-1	40.6	17.8	32.1	9.5
TPSS-2	33.6	16.4	35.4	14.6
TPSS-3	33.5	12.7	41.0	12.8
TPSS-4	19.1	29.8	36.1	14.9
TPSS-5	37.5	21.5	27.6	13.3
TPSS-6	11.9	12.6	51.8	23.7
TPSS-7	43.3	30.4	17.8	8.6
TPSS-8	28.2	23.1	37.2	11.5
TPSS-9	58.8	29.4	8.2	3.6
TPSS-10	62.2	25.0	9.2	3.7
TPSS-11	52.8	23.5	16.4	7.3

* British Soil Classification System (BS 5930:1981)

TABLE 3.5: GRAINSIZE ANALYSIS RESULTS. GRAINSIZE DISTRIBUTION CURVES ARE GIVEN IN APPENDIX 3.

Permeability Testing

Following work by Davidson Partners Ltd (DPL) on cover material permeability testing and their recommendation of testing partially compacted cover materials by the constant head permeability method due to their free flowing nature, 4 test cells were set up following British Standards BSI 1377:1990. The British Standards recommend that grains greater than 4 mm be removed from the test sample prior to testing. The resulting samples once partially compacted, in contrast to DPL results, proved too impermeable upon saturation to produce measurable flow and thus unable to be further tested by the constant head method, implying sample permeability of less than 10^{-5} g/m^3. No reference is given by DPL to either sample preparation or testing method, and it is possible that samples prepared and tested by DPL may have included grain sizes over 4 mm in diameter which, when reconstituted into a test permeameter, significantly increases the measured permeability.

In response to the failure of constant head testing undertaken by the author, eleven cover material samples were tested using the falling head method (refer Appendix 3). Samples were dry sieved to remove grains greater than 4 mm as discussed above. The removal of coarse grain sizes during sample preparation may have the effect of decreasing measured permeability, especially in those samples with high gravel content.

Adjusted samples were then partially compacted into the test cells; standard compaction methods were avoided so as to minimise damage to test equipment and to ensure samples were not fully compacted (refer Appendix 3.6 for compaction method). Samples were immersed in water and left for 24 hours to saturate, with cells checked regularly to ensure no leakage. Measurement by the falling head method proceeded 24 hours after immersion, and water levels were tested at regular intervals based on the rate of head drop for each individual sample. At this time, only samples TPSS-1, 2, 3 and 6 could be measured; the drop in hydraulic head of the other seven samples remained too rapid after 24 hours to permit accurate measurement. The rapid drop in hydraulic head suggests an unsaturated hydraulic conductivity in excess of 10^{-6} m/s. All samples were measured over an 8-hour period, 48 hours after being immersed in water. The different values for permeability after different times reflects the effect of air filled voids and the consequent change in hydraulic conductivity with time during the saturation process. Measured permeabilities after both 24 and 48 hours are given in Table 3.6

Results give permeabilities up to two orders of magnitude greater than those obtained by DPL, ranging from a minimum permeability of 1.76×10^{-8} m/s for TPSS-6 to a maximum of 4.65×10^{-6} m/s for TPSS-8. The spatial distribution of permeabilities indicate that in general, the least permeable materials occur within the capped section of the landfill; the permeability of TPSS-6 reflects the high proportion of silts in comparison to other samples. Overall, the permeability of the cover layer in the capped area comply with resource consent requirements of cover material permeability less than 10^{-7} m/s; samples from the stockpile areas are variable reflecting the variability of stockpiled material; and the remainder of the landfill cover does not comply with resource consent requirements.

	Permeability	
Sample #	After 24 hours	After 48 hours
TPSS-1	2.14×10^{-7}	6.66×10^{-8}
TPSS-2	6.69×10^{-8}	2.34×10^{-8}
TPSS-3	1.10×10^{-7}	2.55×10^{-8}
TPSS-4	*	1.39×10^{-6}
TPSS-5	*	6.86×10^{-7}
TPSS-6	4.89×10^{-7}	1.76×10^{-8}
TPSS-7	*	2.97×10^{-6}
TPSS-8	*	4.65×10^{-6}
TPSS-9	*	1.95×10^{-6}
TPSS-10	*	4.83×10^{-7}
TPSS-11	*	3.99×10^{-6}

TABLE 3.6: EXISTING LANDFILL COVER SOIL – FALLING HEAD PERMEABILITY RESULTS. .

3.6.3 Landfill Geometry

Verbal accounts of gravels excavation at the Taylor Pass Landfill indicate that the base of the excavation coincided with groundwater level at the time of excavation and followed the extent of the top surface of the "blue pug" layer aforementioned. The excavation base was near flat, mimicking the gradient of the surrounding river terraces (Jim Dovey, 1999).

Six trenches were excavated around the perimeter of the landfill site in order to locate the landfill base (see Trench Logs in Appendix 2). The base heights from the trenches were projected to through the landfill and the depth of the landfill ascertained. Figure 3.15 illustrates the geometry of the Taylor Pass Landfill. Isopachs illustrated in Figure 3.15 are based on the correlation of projected landfill basal contours and surveyed upper surface contours south of the Composting Centre. No topographical survey has been carried out on the northern part of the landfill and surface contours are assumed based on field investigations and aerial photo interpretation. Inferred topography of the northern portion is expected to be correct to within 1-2 m.

Based on isopachs shown in Figure 3.15 the volume of the Taylor Pass Landfill has been calculated at approximately 1.6×10^6 m^3 (Appendix 3). The corresponding mass of waste remains undetermined due to the variability in compaction rate across the site. Prior to 1993 little or no compaction of waste was carried out, and although a compactor was introduced to the site in 1993 and used until closure in 1996, there is no record of the extent to which compaction of refuse was carried out over this period. Although the recommended compaction of refuse is between 0.8 and 1.0 t/m^3 (CAE, 1992) there is no indication that this density was achieved in the latter years of

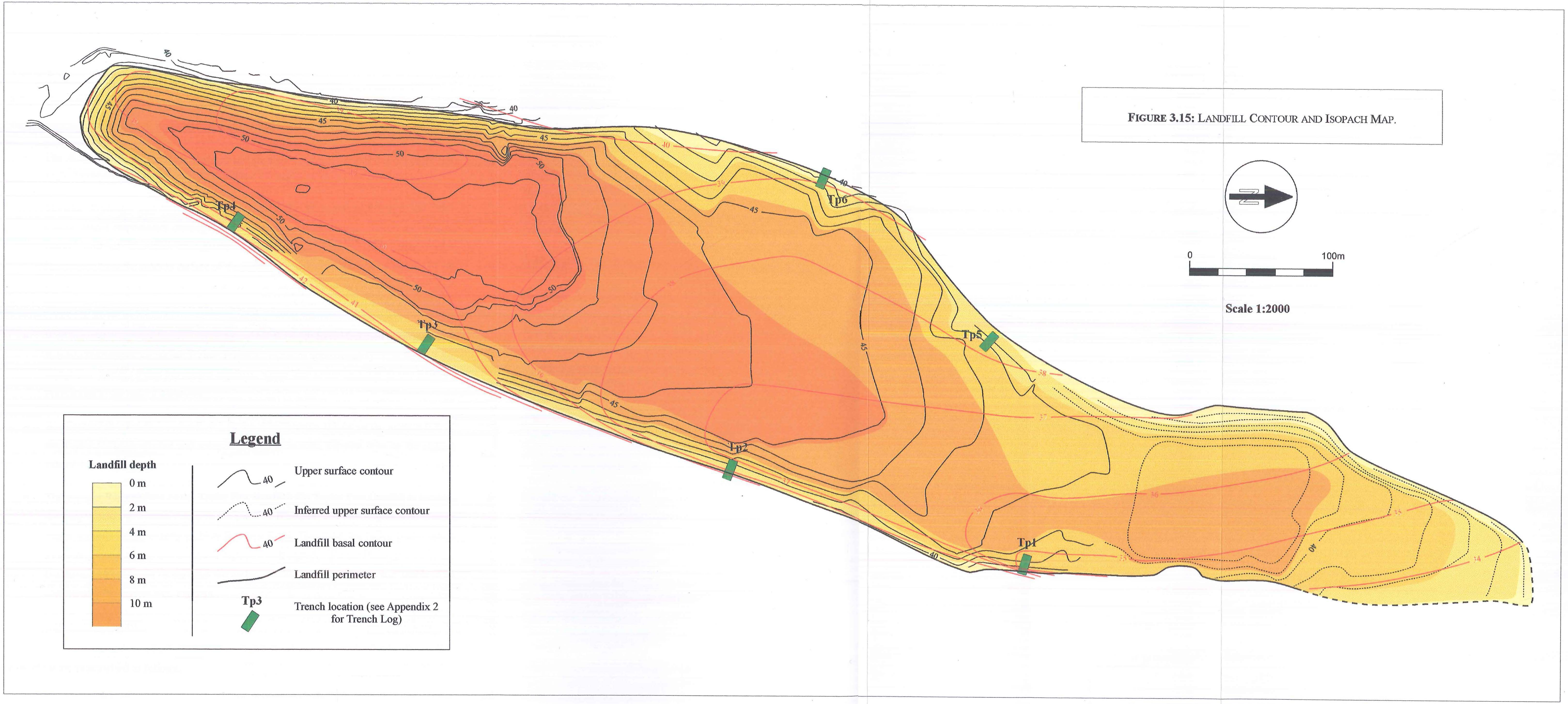


FIGURE 3.15: LANDFILL CONTOUR AND ISOPACH MAP.

landfill operation, and certainly refuse disposed prior to the introduction of a compactor will be significantly less dense.

3.7 Chapter Summary

3.7.1 Geology

The geology and srtatigraphy of the Wairau Valley is summarised as follows:

- The Wairau Plains comprise a series of glacial and interglacial deposits infilling a fault angle depression formed by the Wairau Fault.

- Manuka, Tophouse and Speargrass Formations of the Waimaunga, Waimea and Otira Glacial stages respectively outcrop in the southern tributary valleys and on the Wairau Plains, west of Renwick. Post-glacial (<14000 years) Rapaura and Dillons Point Formations form the modern surface of the Wairau Plains east of Renwick. Deposits of the Karoro interglacial period which separate the Manuka and Tophouse Formations have been tentatively identified by MDC (1998) at approximately 85 m deep in the Bells Road area.

- Well log correlations by Brown (1981a,b) and MDC (1998) have identified Speargrass and Rapaura Formations across the Wairau Plains to depths of approximately 30 m through the mid-plains. Dillons Point Formation separates upper and lower sections of the Rapaura Formations to the east of Blenheim.

- The Taylor Fan comprises two main lobes; the older Speargrass lobe on the western side of the Taylor Valley is eroded and redeposited by the modern Rapaura lobe on the eastern side of the valley.

- The younger Rapaura lobe of the Taylor Fan, in which the Taylor Pass Landfill is located, discharges onto the Wairau Plains to the south of Blenheim and interfingers with Wairau Plains deposits in the vicinity of New Renwick Road. The distal fan facies grades into a thin fine grained swamp and overbank deposit which caps the Wairau Plains gravels in the Blenheim area. Deposits of the main fan surface are highly discontinuous and unable to be correlated using current well log data.

3.7.2 Taylor Pass Landfill

The Taylor Pass Landfill is located within the modern flood channel of the Taylor River. Site conditions are summarised as follows:

- The landfill is capped at the southern end with sandy clayey silt and gravel. The remainder of the landfill that isn't landscaped or occupied by tracks is covered by clayey silt with gravels and sands to sandy gravel with silt and clay.

- The depth of cover ranges to at least 2.1 m with an approximate average depth of 0.7 m. Thirty percent of the landfill is adequately vegetated with wattle and pine, whilst a significant portion of the landfill remains inadequately vegetated. An offal pit located in the central landfill area, together with associated excavated refuse, occupy approximately 1 ha.

- The landfill covers a total of 23.8 ha and reaches a maximum depth of 10-11 m in the southern section then thins to the north. The total volume of the landfill is of the order of $1.6 \times 10\ m^3$.

Chapter 4

Hydrology and Hydrogeology

4.1 Introduction

Groundwater, its origin and its movement are investigated in this chapter at a number of scales. Wairau Plains hydrology and hydrogeology is firstly introduced to give a regional-scale picture of the depositional environment of Wairau aquifers, and more importantly the groundwater flow characteristics. The Taylor Fan Aquifer is then investigated and discussed with respect to flow patterns and constraining geological factors and hydraulic characteristics. On a smaller scale still, water budget analysis and hydrogeological investigations of the Taylor Pass Landfill aim to assess the influence of meteoric and groundwater-derived sources for leachate generation at the Taylor Pass Landfill.

4.2 Wairau Plains Aquifer System

4.2.1 Surface Waters

Flowing down the northern margin of the Wairau Plains, the Wairau River flow is sourced primarily from the Upper Wairau Valley. The Opawa River (Figure 4.1) flows down the southern margin of the Wairau valley and represents the former southern distributary of the Wairau River. Considerable amounts of river control works associated with both the Opawa and Wairau Rivers (Figure 4.1) have been aimed at reducing the flood risk to Lower Wairau Plains communities. For details of flood control works on the Wairau Plains, the reader is referred Rae (1987) and the Marlborough District Council Floodways Management Plan (1994).

In effect, the river control works have distributed flow of the Southern Tributary Rivers into 3 primary channels meeting at the southern mouth of the Wairau River. The Wairau River takes flow from the Northern tributaries, and southern tributaries west of, and including, the Waihopai River; the upper Opawa and Roses Overflow take flow from the Omaka and Fairhall Rivers and Mill Stream; and the Lower Opawa River takes flow from Golf Course and Doctors Creeks and the Taylor River (Figure 4.1). The extent of works undertaken is greater than any other flood control

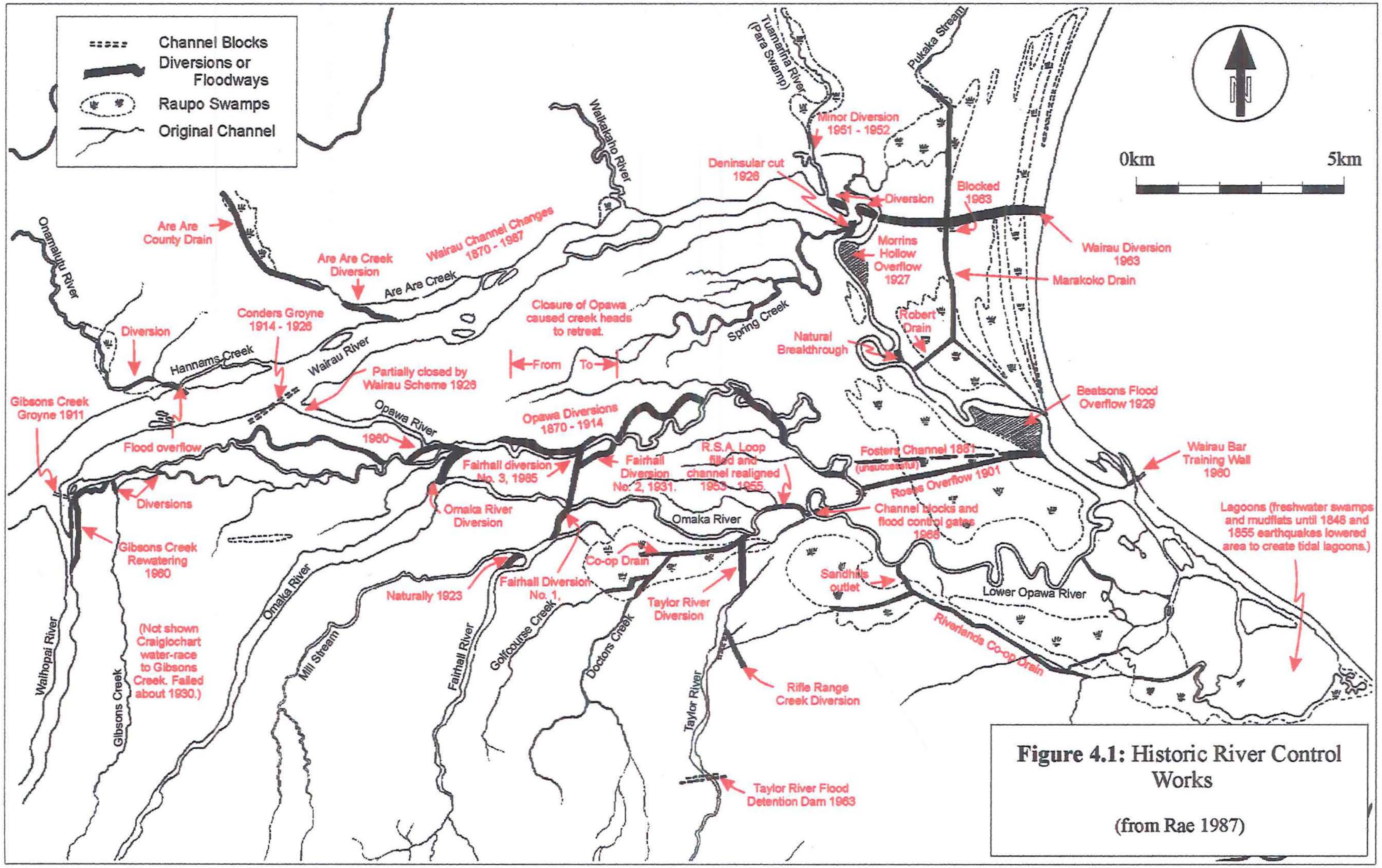


Figure 4.1: Historic River Control Works

(from Rae 1987)

works in New Zealand and reflect the fact that the rivers of the area are highly susceptible to flooding and consequent change of course. Thus over the period of deposition of both Rapaura and Speargrass Formation alluvial gravels, channels are likely to have been very complexly and rapidly migrating, and prone to depositing thin sheet deposits which in turn are further cut by migrating channels, thus leading to laterally discontinuous gravel, silt and sand deposits.

4.2.2 Groundwater

Introduction

The Wairau Plains aquifers are defined by glacial and interglacial sequences as discussed in Chapter 3. Investigations to date have identified the Main Wairau Aquifer comprising the Speargrass and Rapaura Formations as being the most laterally extensive, underlying an approximate area of 15,000 hectares (Davidson *et al.,* 1994). The majority of wells in the Wairau Plains area, including Blenheim municipal water-supply wells, draw water from the Main Wairau Aquifer. Consequently the resource and associated geology are better understood than deeper aquifers, which remain largely undefined with respect to hydraulic characteristics. The following section discusses properties of the Main Wairau Aquifer only, as it is the most vulnerable with respect to contamination from the Taylor Pass Landfill.

Davidson *et al.* (1994) have defined the (Main) Wairau Aquifer as:

> *"the groundwater resources bounded by the Wairau River, Cloudy Bay coast, Waihopai River and New Renwick Road."*

The aquifer is confined to the east of Blenheim by Dillons Point Formation sediments and is unconfined to the west. The boundary between confined and unconfined conditions is gradual and irregular, forming a partially confined zone in the vicinity of Blenheim Township (Figure 4.2). The Main Wairau Aquifer is vertically confined at the base by silt and clay deposits of the Kaihinu interglacial stage *circa.* 70,000 – 120,000 BP (refer Chapter 3).

Recharge of the Main Wairau Aquifer occurs largely from the Wairau River with smaller yet still notable recharge occurring from southern tributary rivers and streams. Predominant lateral flow within the aquifer is in an easterly direction towards Cloudy Bay (Figure 4.2). Vertical flow in the Wairau Aquifer is downwards in the unconfined recharge zone, gradually changing through horizontal to upwards flow in the partially confined and confined zones to the east (Figure 4.3). Springs occur in a band through the central Plains area through to Blenheim, representing the boundary between the confined and unconfined zones.

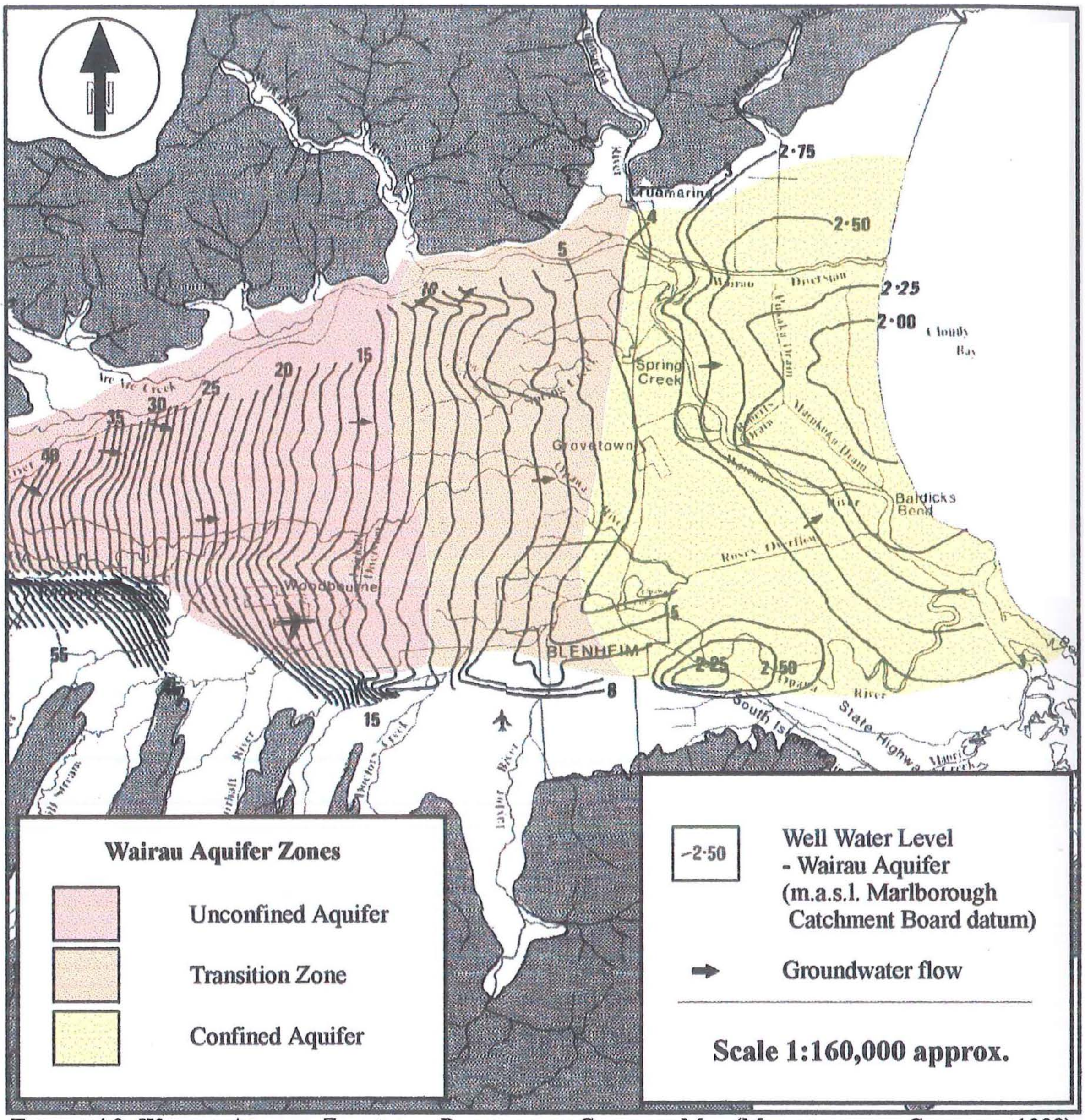


FIGURE 4.2: WAIRAU AQUIFER ZONES AND PIEZOMETRIC CONTOUR MAP (MODIFIED FROM CUNLIFFE, 1988).

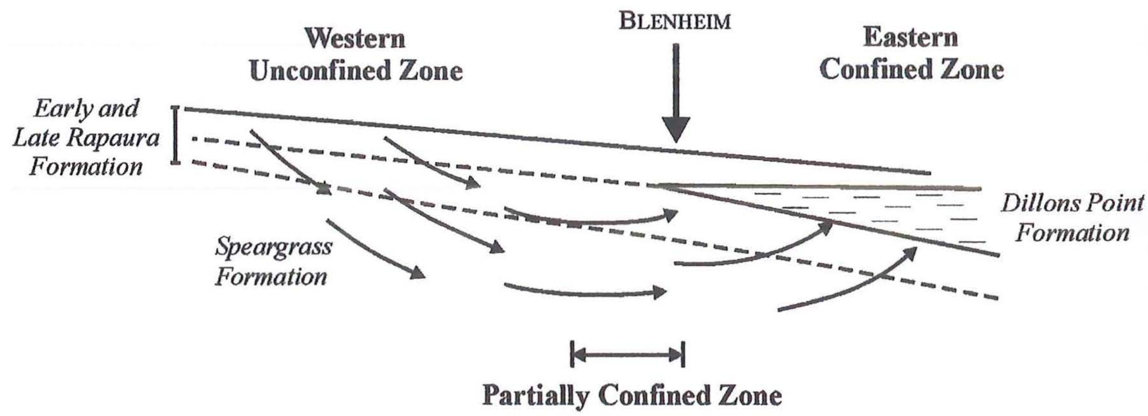


FIGURE 4.3: SIMPLIFIED MODEL OF VERTICAL FLOW IN WAIRAU AQUIFER.

Main Wairau Aquifer Hydraulic Characteristics

The definition of the Wairau Aquifer provided by Davidson *et al.* (1994) above excludes southern valley water resources on the basis of their relative low yield in comparison to the extensive Wairau Aquifer. The Main Wairau Aquifer comprises deposits of the Speargrass and Rapaura Formations (Section 3.3), with the reworked and hence cleaner Rapaura gravels offering the highest yield.

Figure 4.4 illustrates the regional trend of specific capacity of wells across the Wairau Aquifer. High yielding wells (> 400 m^3/hour/m) are located adjacent to the middle reaches of the Wairau River, with decreasing well yields towards the southern tributary valleys. At the periphery of the Main Wairau Aquifer, specific capacity values range from 10 to 100 m^3/hour/m; specific capacities of less than 10 m^3/hour/m define the boundary of the southern tributary aquifer systems.

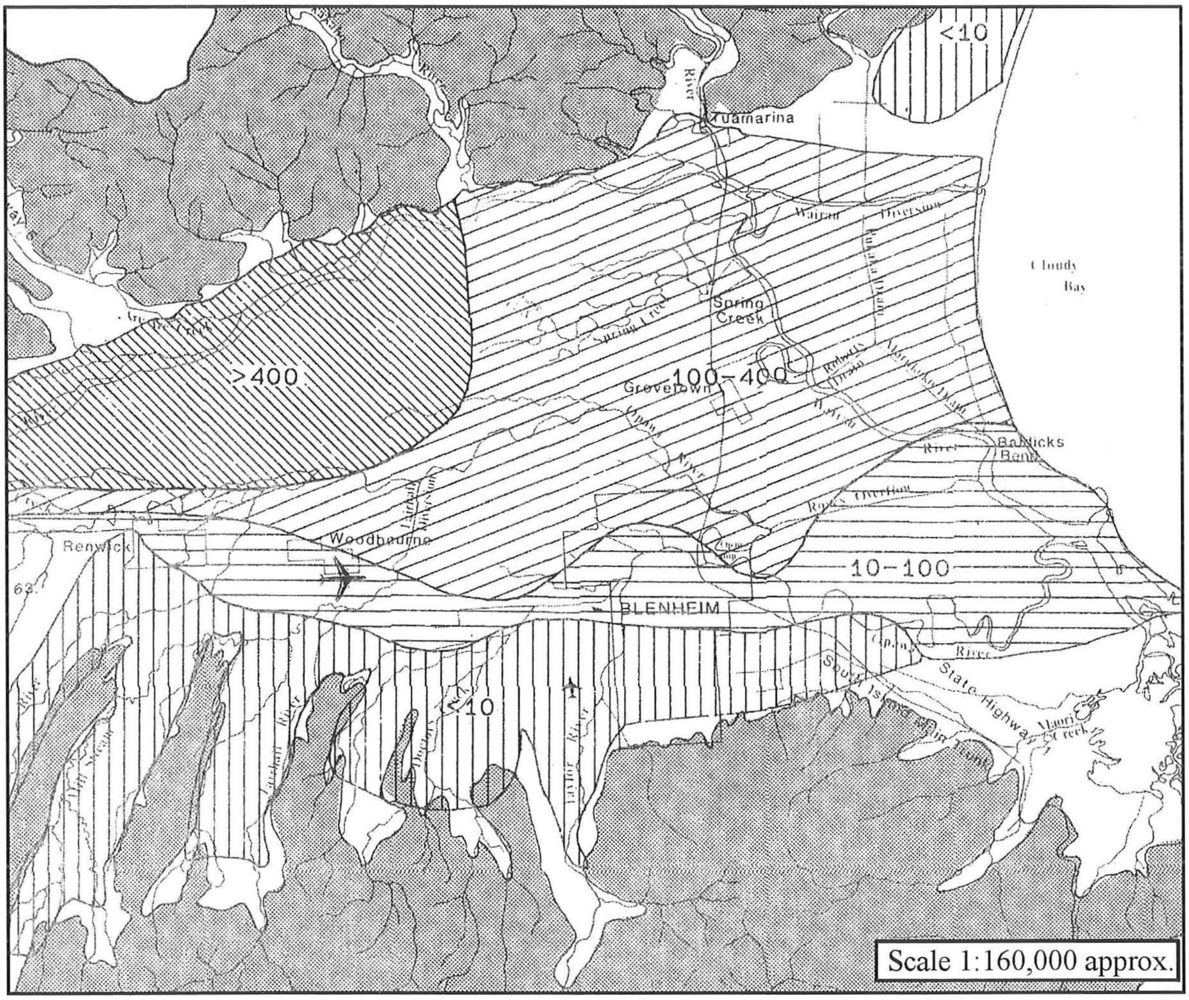


FIGURE 4.4: SPECIFIC CAPACITY DATA ACROSS THE WAIRAU PLAINS IN M^3/HOUR/DAY (FROM RAE, 1988).

Transmissivity values of the Main Wairau Aquifer based on step-drawdown well pumping tests with and without observation wells (refer Fetter, 1994) range from approximately 500-15,000 m^2/day (Davison *et al.*, 1994). Transmissivities in the area of interest (Blenheim) range from 1000 to 13,000 and are typically noted between 3000 and 7000 m^2/day. Davidson *et al.*, suggest that the variation in observed transmissivity values may be attributable to partial penetration of testing wells and changes in unconfined aquifer levels, however the seasonal change in aquifer levels over the majority of the Lower Wairau Area is less than 1 metre and up to 2 m at the aquifer boundary. The exception to this seasonal variation is in the Woodbourne area west of the Taylor Fan where seasonal fluctuations of over 8 m have been observed (Figure 4.5).

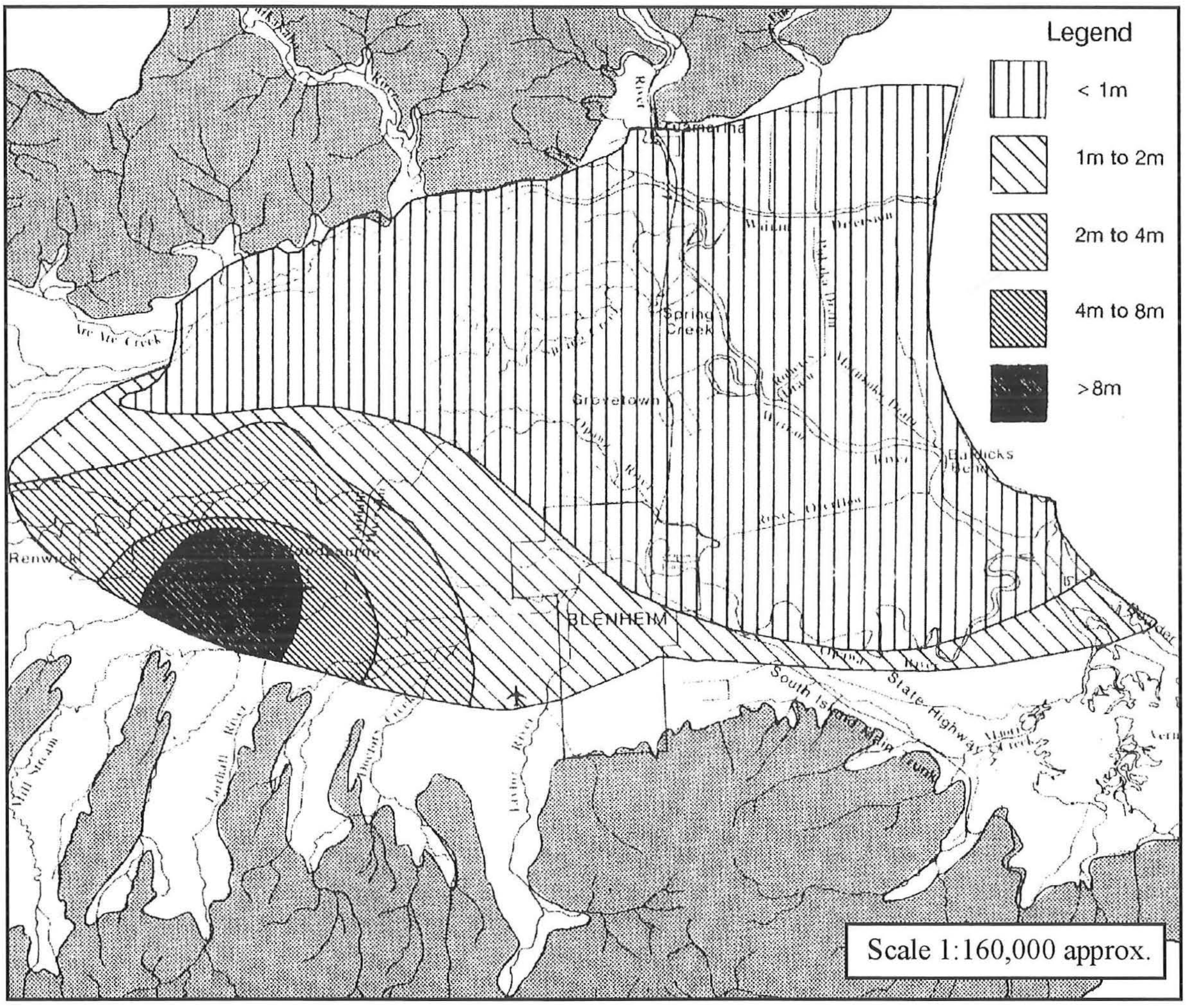


FIGURE 4.5: SEASONAL VARIATION IN GROUNDWATER LEVEL IN THE WAIRAU AQUIFER (FROM CUNLIFFE, 1988).

Based on transmissivity values of 1000 to 13 000 m^2/day in the Blenheim area, a hydraulic gradient of 0.0023 given by Davidson *et al.*, and assuming an aquifer thickness of 30 m (see Figure 3.14) and effective porosity of 35% for Rapaura deposits, groundwater flow velocities of the Main Wairau Aquifer in the Blenheim area range from 0.2 to 2.8 m/day. Variability of groundwater flow velocity is a result of preferential flow paths defined by buried channels and other predominantly coarser grained layers.

Storativity values across the Main Wairau Aquifer reflect the nature of the aquifer's upper bounding surface. Unconfined aquifer storativity values of the order of 10^{-1} decrease through the semi-confined zone (10^{-3}) to values of the order of 10^{-5} in the confined aquifer zone (Davidson *et al.*, 1994). Storativity values of 10^{-4} to 10^{-5} observed in the Blenheim area suggest confinement of the Main Wairau Aquifer in this area. This confirms the presence of and especially the continuity of a layer of fine sediments possibly associated with the distal fan facies of the Taylor Fan at or near the surface through the Blenheim area (as discussed in Chapter 3). Whether this confining layer is connected to or associated with the Dillons Point Formation confining layer remains undetermined.

The nature of interaction and boundary between the Wairau and Taylor Fan Aquifers has been discussed in Chapter 3 and is further discussed in Section 4.4 and Chapter 5 in an effort to clarify the risk of contamination of Wairau groundwaters from the Taylor Pass Landfill. The lateral boundaries of deeper aquifers defined by Marlborough District Council (1998) have not been considered in this thesis.

4.3 Taylor River

The Taylor Catchment covers an area of approximately 70 km^2 stretching south from Blenheim towards the Awatere Valley. The Taylor River has proved historically to be a great flood risk to the Blenheim Township, often filling or overtopping its meandering channel through the town centre. Figure 4.6 gives an indication of the erratic nature of flow in the Taylor Catchment. Since 1962, 75% of average daily flow rates have remained below 500 l/s, yet over the same period the maximum average daily flow reached up to 96,108 l/sec.

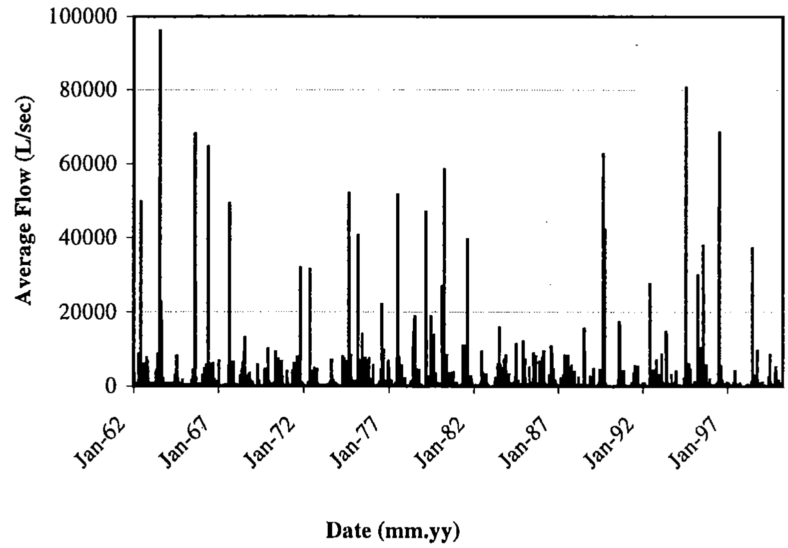


FIGURE 4.6: TAYLOR RIVER DAILY AVERAGE DISCHARGE AT BOROUGH WEIR (GRID REF. P28/879565) - 1962 – 2000 (DATA FROM MDC).

In 1964-65 a flood detention dam was built approximately 5 km upstream from New Renwick Road to reduce the risk of flooding in Blenheim. After modification to construction following review in 1980, the dam has a design capacity of 311.5 cumecs with a reduced outflow culvert of 113.3 cumec capacity (Marlborough Catchment Board, 1988).

Dam construction also reduced the risk of excessive scouring of the river banks during flood events, which otherwise may have led to further environmental problems associated with the remobilisation of solid wastes from both the Taylor Pass and Brayshaw Park Landfills. Downcutting of up to 3 m in the Taylor River bed has occurred since dam construction to 1991 (Figure 4.7), however, most likely as the result of reduced sediment supply.

Adjacent to the Taylor Pass Landfill flows remain seasonal and sporadic, but no accurate flow data exists. When surface flow does occur, the Taylor River is likely to act as a recharge source to the unconfined Taylor Aquifer System, although, river flow gauging between the Taylor Dam and New Renwick Road is required to clarify this. During dry periods, flow through the section between the Taylor Dam and New Renwick Road section of the Taylor River is likely to be represented by underflow within the gravels.

4.4 Taylor Fan Aquifer

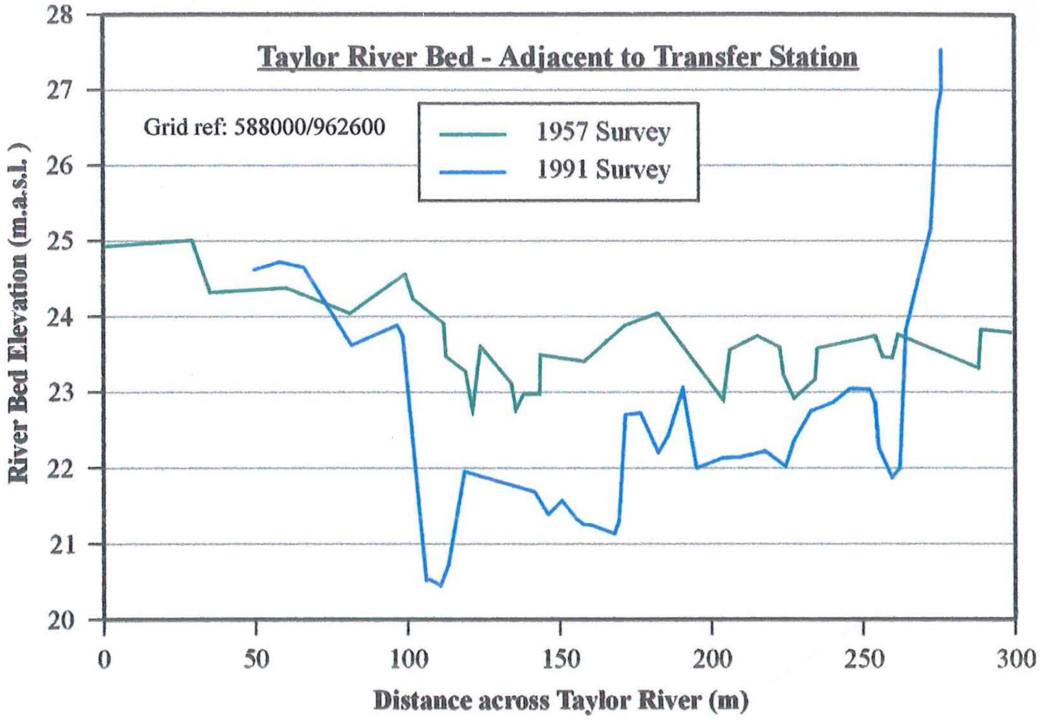
4.4.1 Investigation Background

As discussed in previous chapters, the lack of demand for groundwater in the Taylor Pass area and the relative abundance of supply in the Wairau Aquifer have resulted in a lack of interest in the Taylor Fan Aquifer as a viable groundwater resource with the exception of the distal fan area north of New Renwick Road. Consequently, no exclusive hydrogeological investigations of the main Taylor Fan surface (south of New Renwick Road) have been carried out to date.

4.4.2 Investigation Methodology and Limitations

Introduction

The present investigation of the Taylor Fan Aquifer is aimed at establishing the nature, rate and direction of groundwater flow within the aquifer and its connection with the Main Wairau Aquifer. Direct field methods used commonly to determine such parameters include extensive piezometric surveys and short and long-term pumping tests of wells across the investigation area. The availability and distribution of wells, and the quality and relevance of well test data, are critical in establishing and verifying aquifer properties.



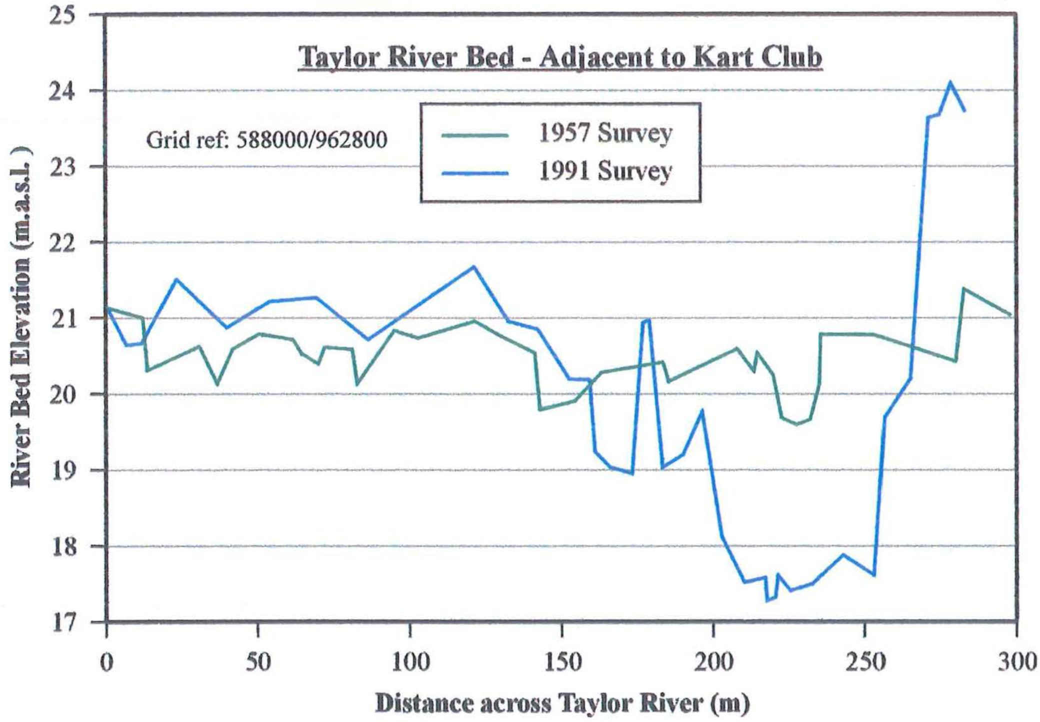


FIGURE 4.7: COMPARISON OF THE TAYLOR RIVER BED PROFILE, 1957 AND 1991. SECTIONS ARE VEIWED IN A DOWNSTREAM DIRECTION. VERTICAL EXAGGERATION 1:25.

Well Availability

There are significantly more wells on the distal region of the Taylor Fan surface than indicated in Chapter 3 as those having lithological logs. Many abandoned domestic supply wells concentrated in the Redwoodtown area, however, are inaccessible for direct testing or measurement due to the nature or location of the well head (e.g. buried and/or bowed and/or permanently attached to domestic supply or irrigation pumps). A number of wells in the Aerodrome Road and Burleigh areas are subject to the same difficulties.

West of the Taylor River, with the exception of P28/W3390, little information on the movement of water within the Speargrass lobe of the Taylor Fan is available. Immediately north east of the Taylor Pass Landfill, monitoring network wells extend in a rough line from the end of the landfill towards Blenheim, giving good control of the change in hydraulic gradient through the Rapaura Formation Lobe of the Taylor Fan in a northeasterly direction but with little lateral control.

The location of accessible wells over the Main Taylor Fan places a bias on the detail of piezometric surfaces and results in the lack of lateral control necessary to accurately resolve flow patterns (refer Section 4.4.4). Although this problem is common to all hydrogeological investigations, it is especially important to note when dealing with the movement and extent of a potential contaminant plume.

Aquifer Test Restrictions

Although considered necessary in the investigation of aquifer properties, long duration pump tests (i.e. in excess of 24 hours) were not a feasible option for testing of hydraulic characteristics because of:

1. Serious logistical problems associated with the disposal of contaminated or potentially contaminated water discharged over the duration of a long-term pump test.

2. Difficulties with fitting a suitable submersible pump and water level monitoring equipment in the small diameter wells (maximum diameter wells of 75 mm) located on the Rapaura surface of the main Taylor Fan.

3. Noise problems associated with long term pumping of wells in or near established residential areas.

Single well slug tests (refer to Fetter, 1994, for methodology) were thus identified as a feasible alternative investigation method for the assessment of hydraulic characteristics in the Taylor Fan area as none of the aforementioned problems associated with long term pump testing need be applicable to the slug test method.

Slug tests were attempted on small diameter (50 mm and 75 mm) Taylor Pass Landfill monitoring network wells by inserting a slug, in the form of a weighted aluminium tube, to displace a known volume of water and measuring recovery back to the original groundwater level. The slug was then removed, effectively removing a known volume from the well and once again measuring recovery of the piezometric level back to its original state.

The slug test method proved of little use in the area due to the effect of sand filters around and extending above the screened length of the small diameter wells. The effect of the filter is to dissipate water within the permeable sand before the effects of the surrounding material could be ascertained. This trend is especially problematic in small diameter wells where the volume of packing filter is proportionally large compared to that of the well screen and casing. For example, in monitoring wells with a casing diameter of 75 mm, the effective radius of the well casing and packing material is 50 mm, which corresponds to a 78% increase in effective volume. Rapid displacement of sufficient water within the well to overcome the initial 56% loss to the filter materials was not viable, and slug testing was therefore also abandoned as a method of investigation.

Investigation Methodology

Following initial difficulties in investigation methods as discussed above, hydrogeological characterisation of the Taylor Fan Aquifer has been based on data from the following sources, which are further discussed in subsequent sections:

- short duration single well yield-drawdown test results carried out immediately following construction of some household supply wells in the area, which were obtained from the MDC Well Log databank;

- extrapolation and interpretation of results of long duration hydraulic testing in similar depositional environments on neighbouring southern tributary fans

- a spatially restricted piezometric survey carried out in June 1999, and comparison with static water levels in selected wells measured regularly as a part of the monitoring regime for the Taylor Pass Landfill;

- continuous water level records from a pressure transducer and data recorder installed at well P28/W3387 over the period from April 1999 to April 2000;

4.4.3 <u>Hydraulic Characteristics</u>

Estimated Transmissivity from Specific Capacity Data

Short duration well pumping test results in the MDC databank consist of:

- Pumping rate,
- Pumping duration, and
- Final drawdown.

The specific capacity of a well is defined as the yield of the well divided by the drawdown, and thus is obtainable from short duration pump tests. Appendix 4 outlines how the specific capacity of a well can be used to estimate the corresponding transmissivity of the aquifer. In addition to generic assumptions listed in Appendix 4, it must be emphasised that single well pump tests measure the properties of the aquifer only in the close vicinity of the testing well and may not be representative of the aquifer as a whole. Figure 4.7 illustrates the distribution of transmissivities based on specific capacity data. Spatial distribution of specific capacity data is dictated by domestic groundwater use, and is concentrated therefore at the boundary between the main Wairau Aquifer and the distal Taylor Fan north of New Renwick Road.

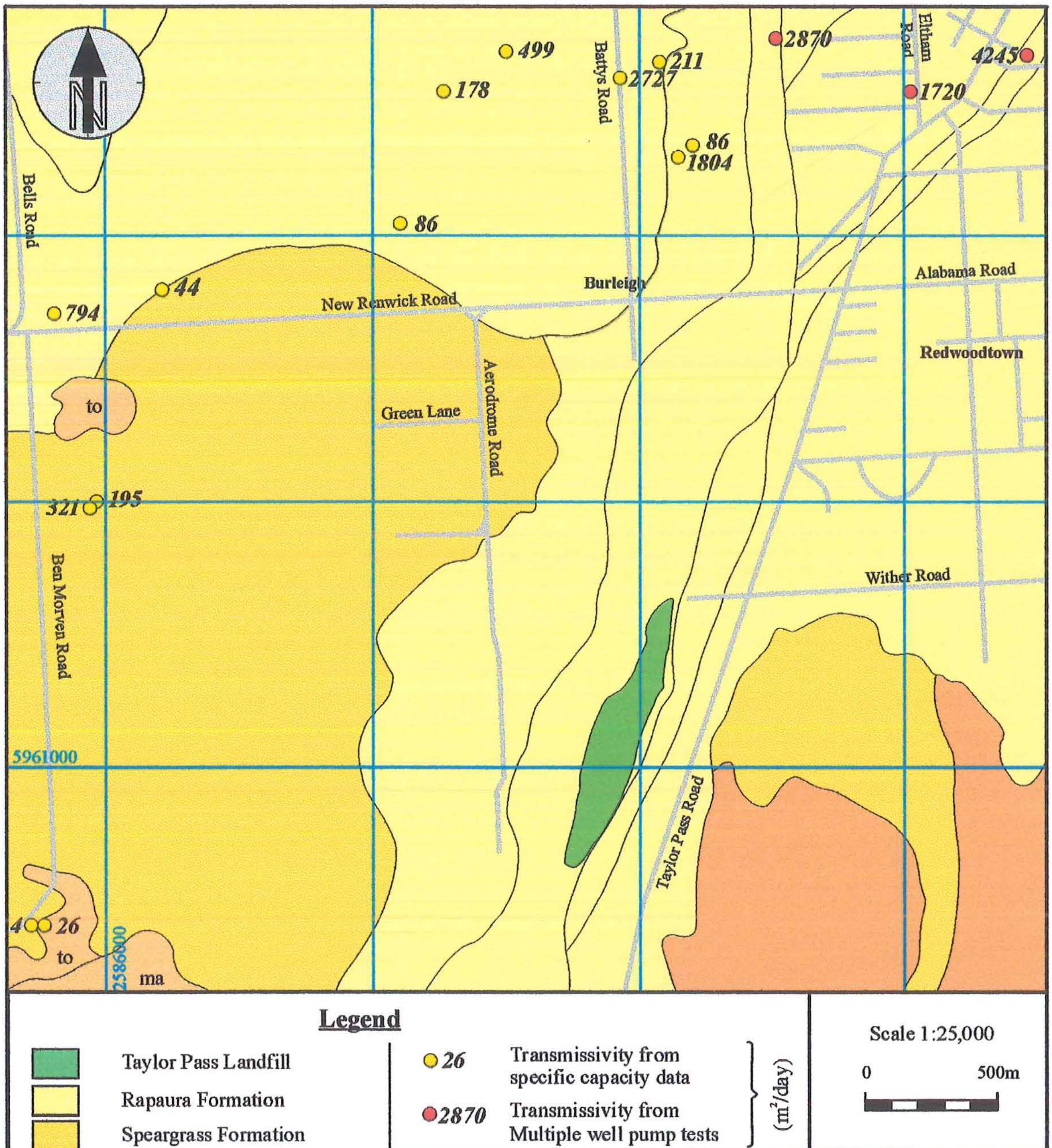


FIGURE 4.8: TAYLOR FAN TRANSMISSIVITY FROM SPECIFIC CAPACITY DATA AND MULTIPLE WELL PUMP TESTS

Table 4.1 lists corresponding hydraulic conductivities based on the effective depth of the pumping well. The effective depth of the pumping well within a confined aquifer is considered to be from the base of the screened interval to the base of the confining layer. Where an aquifer is unconfined, the effective depth is assumed to be equal to the distance from the static water level to the base of the screened interval. A well is considered to penetrate a confined aquifer if the water level over the screened interval rises above the level of an overlying clay rich layer acting as a barrier to vertical flow. For transmissivity calculations, storativity has been estimated at 0.1 for unconfined aquifers and 0.0001 for unconfined aquifers. Calculation sheets for both transmissivity and hydraulic conductivity can be found in Appendix 4.

Well Number	Grid Ref	Aquifer nature	Formation	Transmissivity (m^2/day)	Hydraulic Conductivity (m/s)
P28/W1225	6200/3800	Confined	Speargrass	44	5.2×10^{-05}
P28/W1384	7931/4600	Confined	Rapaura (ra_1)	2727	1.3×10^{-02}
P28/W1663	8080/4660	Unconfined	Rapaura (ra_1)	134	2.5×10^{-04}
P28/W1784	5790/3710	Confined	Rapaura (ra_1)	794	1.3×10^{-03}
P28/W2090	5900/3000	Confined	Speargrass	195	5.6×10^{-04}
P28/W2511	7500/4700	Unconfined	Rapaura (ra_1)	303	2.7×10^{-04}
P28/W2566	8150/4300	Unconfined	Rapaura (rg)	1804	1.4×10^{-03}
P28/W2879	5926/2980	Confined	Speargrass	321	4.6×10^{-04}
P28/W2885	5750/1400	Confined	Speargrass	26	8.7×10^{-05}
P28/W2949	7100/4050	Confined	Rapaura (ra_1)	86	2.3×10^{-04}
P28/W3177	5697/1417	Confined	Speargrass	4	7.2×10^{-07}
P28/W3228	8204/4347	Unconfined	Rapaura (rg)	86	9.7×10^{-05}
P28/W3277	7265/4549	Unconfined	Rapaura (ra_1)	178	2.3×10^{-04}

TABLE 4.1: TRANSMISSIVITY AND HYDRAULIC CONDUCTIVITY OF RAPAURA AND SPEARGRASS DEPOSITS OF THE DISTAL TAYLOR FAN-WAIRAU AQUIFER BOUNDARY ESTIMATED FROM SPECIFIC CAPACITY DATA.

Multiple Well Pumping Tests.

The Marlborough District Council (1991) carried out short duration multiple-well pumping tests of supply bores at Eltham Road (P28/W1313), Athletic Park (P28/W949) and Graham Street (P28/W723) in 1984 to evaluate the groundwater resource potential (refer red wells in Figure 4.7). Table 4.2 outlines the transmissivity and storativity results of testing as published in Cunliffe (1988). Hydraulic conductivities have been further calculated based on the aquifer thickness interpreted from well log data. All wells are located within Rapaura Formation gravels.

Measured storativity values of the three wells indicate low elasticity in a confined aquifer system. This correlates well with the identification of a confining layer associated with distal fan deposits as discussed in Chapter 3, and suggests that the layer is indeed continuous between these wells at least.

Well Number	Location	Aquifer thickness (m)	Transmissivity (m^2/day)	Storativity	Hydraulic Conductivity (m/s)
P28/W0723	Grahams Road	15.8	4245	0.000073	3.1×10^{-3}
P28/W0949	Athletic Park	61.6	2870	0.000049	5.4×10^{-4}
P28/W1313	Eltham Road	16.2	1720	0.000052	1.2×10^{-3}

TABLE 4.2: TRANSMISSIVITY, STORATIVITY AND HYDRAULIC CONDUCTIVITY OF RAPAURA FORMATION GRAVELS FORM MULTIPLE-WELL PUMP TESTS (DATA FROM CUNLIFFE, 1988).

Hydraulic Characteristics of Other Southern Tributary Aquifers

The Benmorven, Brancott (including Fairhall River gravels), Omaka and Omaka River Valley, and Lower Waihopai Aquifers (Figure 4.9) make up the southern tributary valley aquifer systems to the west of the Taylor Fan Aquifer. Figure 3.2 (refer Chapter 3) indicates that the surface geology of the neighbouring systems comprises significantly more Speargrass Formation gravels across the main aquifer bodies than in the Taylor Fan Aquifer, with Rapaura gravels occurring only in the proximity of the Main Wairau Valley floor and in modern, discrete channel type aquifers of the Omaka and Fairhall Riverbeds and the Mill Streambed.

For extrapolation purposes then, the area represented by Fairhall River Gravels comprising Rapaura Formation may be considered comparable to the Rapaura Formation gravels that form the modern flood plains and degradation surfaces of the main Taylor Fan Area. Speargrass formation gravels along the western margin of the Taylor Valley extending to and including the Aerodrome area are considered comparable to Speargrass deposits of the Benmorven and Brancott Aquifers.

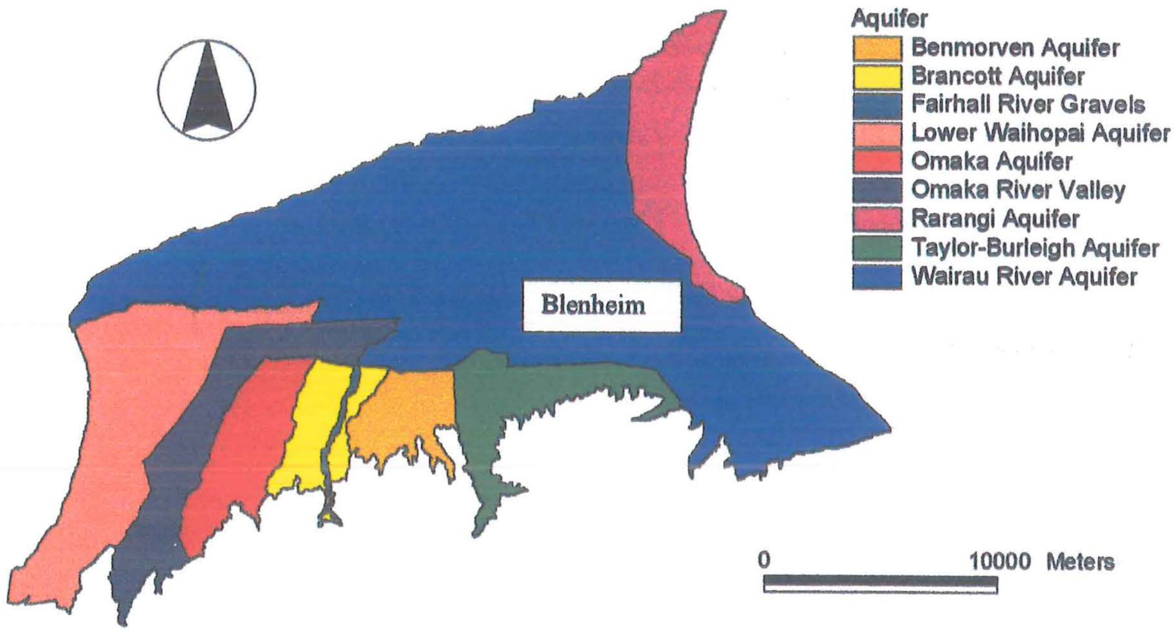


FIGURE 4.9: AQUIFER SYSTEMS OF THE WAIRAU PLAINS. NB. BOUNDARIES ARE THOSE ALLOCATED BY MDC FOR WATER RESOURCE IDENTIFICATION AND MANAGEMENT PURPOSES. THE EASTWARD EXTENT OF THE TAYLOR-BURLEIGH (TAYLOR) AQUIFER INCLUDES SMALL TRIBUTARY FANS EMANATING FROM THE WITHER HILLS AND IS NOT CONSIDERED TO BE PART OF THE TRUE TAYLOR AQUIFER REFERRED TO IN THIS PROJECT.

Transmissivities obtained from long duration pump tests carried out in both Rapaura and Speargrass gravels in southern tributary aquifer systems west of the Taylor Fan System as reported by Cunliffe (1988) are given in Table 4.3. Hydraulic conductivities have been further calculated assuming a nominal aquifer thickness/effective pumping depth of 20 m.

Aquifer System	Formation	Well No.	Transmissivity (m^2/day)	Storativity	Hydraulic Conductivity (m/s)
Benmorven	Speargrass	P28/W0875	7	-	4×10^{-6}
	Speargrass	P28/W0865	9	-	5×10^{-6}
Brancott	Speargrass	P28/W1172	55	-	3×10^{-5}
Fairhall River gravels	Rapaura (fa)	P28/W1211	528	0.27	3×10^{-4}
	Rapaura (fa)	P28/W1108	569	-	3×10^{-4}
	Rapaura (fa)	P28/W1210	1153	-	7×10^{-4}
	Rapaura (fa)	P28/W1209	1395	0.044	8×10^{-4}

TABLE 4.3: TRANSMISSIVITY, STORATIVITY AND HYDRAULIC CONDUCTIVITY VALUES OF SPEARGRASS AND RAPAURA FORMATION GRAVELS IN SOUTHERN TRIBUTARY AQUIFER SYSTEMS (FROM CUNLIFFE, 1988). NB. FA REFERS TO MODERN RIVER GRAVELS OF THE RAPAURA FORMATION (AFTER BROWN, 1981).

Comparison of Results

From specific capacity data (Table 4.1), transmissivity of Rapaura Formation deposits ranges from approximately 80 to 2700 m^2/day, with corresponding hydraulic conductivities of the order of 1×10^{-4} to 1×10^{-2} m/s. Hydraulic conductivity values from both the multiple well pumping tests and those extrapolated from neighbouring tributary fans (Table 4.2 and 4.3) fall within this range. High transmissivity at the Grahams Road (4245 m^2/day) well are reflected in the relatively high hydraulic conductivity value of 3.1×10^{-3} m/s. Low transmissivity (86 m^2/day) and hydraulic conductivity at P28/W3228 (refer Table 4.1) are likely to be limited to a small area and reflect the characteristics of vertical successions with few water bearing layers which are no doubt present in small localised zones over the Rapaura Surface due.

Multiple well pump tests and specific capacity tests in Rapaura deposits have all been located in the Wairau Aquifer and distal Taylor Fan Facies with the exception of P28/W1784, and are not considered entirely representative of mid to upper Taylor Fan deposits. Fairhall River gravels are considered likely to be more representative of Rapaura deposits of the main Taylor Fan and the Taylor Pass Landfill surrounds due to similar depositional environments forming likely comparable channel and flood plain deposits cut into an older fan surface. A more restricted range of transmissivity values from 528 to 1395 m^2/day (Table 4.3) may reflect a lack of data in the Fairhall river gravels, but still indicates the hydraulic conductivity of deposits as 10^{-3} to 10^{-4} m/s.

Speargrass Formation deposits have higher silt and clay content (and hence considerably lower transmissivities) than cleaner Rapaura Formation Gravels, and range from 7 to 55 m^2/day in the Brancott and Benmorven areas up to 321 m^2/day between the Taylor Fan and Doctors Creek. Table 4.4 lists ranges of hydraulic characteristics assumed to be representative of Speargrass gravels on the western margin of the Taylor Valley, and Rapaura gravels in the mid to upper Taylor Fan and the distal Taylor Fan/Wairau Aquifer boundary based on all available data.

Formation	Transmissivity (m^2/day)	Storativity	Hydraulic Conductivity (m/s)
Speargrass	5 - 320	-	$2 \times 10^{-7} - 6 \times 10^{-4}$
Rapaura (mid to upper Taylor Fan)	530 - 1400	0.27 – 0.044	$3 \times 10^{-4} - 8 \times 10^{-4}$
Rapaura (Taylor/Wairau boundary)	80 - 2700	$5 \times 10^{-5} - 7 \times 10^{-5}$	$9.7 \times 10^{-5} - 1 \times 10^{-2}$

TABLE 4.4: CHARACTERISTIC HYDRAULIC PARAMETERS OF THE TAYLOR FAN.

When flowing through deposits of varying hydraulic characteristics, following basic laws of physics, water will preferentially flow along the most permeable path generally comprising channel deposits. Hydraulic conductivity values of 10^{-4} to 10^{-7} m/s for the Speargrass Formation are much lower than Rapaura gravels (estimated to be 10^{-3} to 10^{-4} m/s), and suggest that the Speargrass Formation has the potential to act as a lateral barrier to groundwater flowing through the Rapaura Formation which represents the preferential flow path down of the main Taylor Fan surface. Average flow velocity then is likely to be determined by the highest hydraulic conductivity. Seepage velocities through deposits of the Taylor Fan are discussed in following sections with reference to characteristic hydraulic parameters in Table 4.4.

4.4.4 Local Piezometric Surface

Groundwater Flow

Figure 4.9 shows the location of measured wells and the corresponding best-fit piezometric map of a small-scale well survey carried out in June 1999. Piezometric contour lines indicate a steepening hydraulic gradient down the Taylor Fan Surface from 1:180 adjacent to the Taylor Pass Landfill to a maximum of approximately 1:50 400 m south of the landfill. Constantly flowing springs located immediately south of the landfill (Figure 4.10) effectively reduce the piezometric elevation by approximately 3 m (based on surveying of the standing water height in December 1999), resulting in the gentle gradient through the landfill area. Springs are formed by perching of the water table above clay and silt rich layers representing overbank deposits as discussed in Chapter 3. Although no other springs were located during the course of this investigation, anecdotal evidence suggests a number of springs appear mainly in the banks of the Taylor River and flow sporadically through the year. The piezometric surface through this area then is likely to consist more of a series of steps associated with spring flow and intermittent areas with gentle hydraulic gradients.

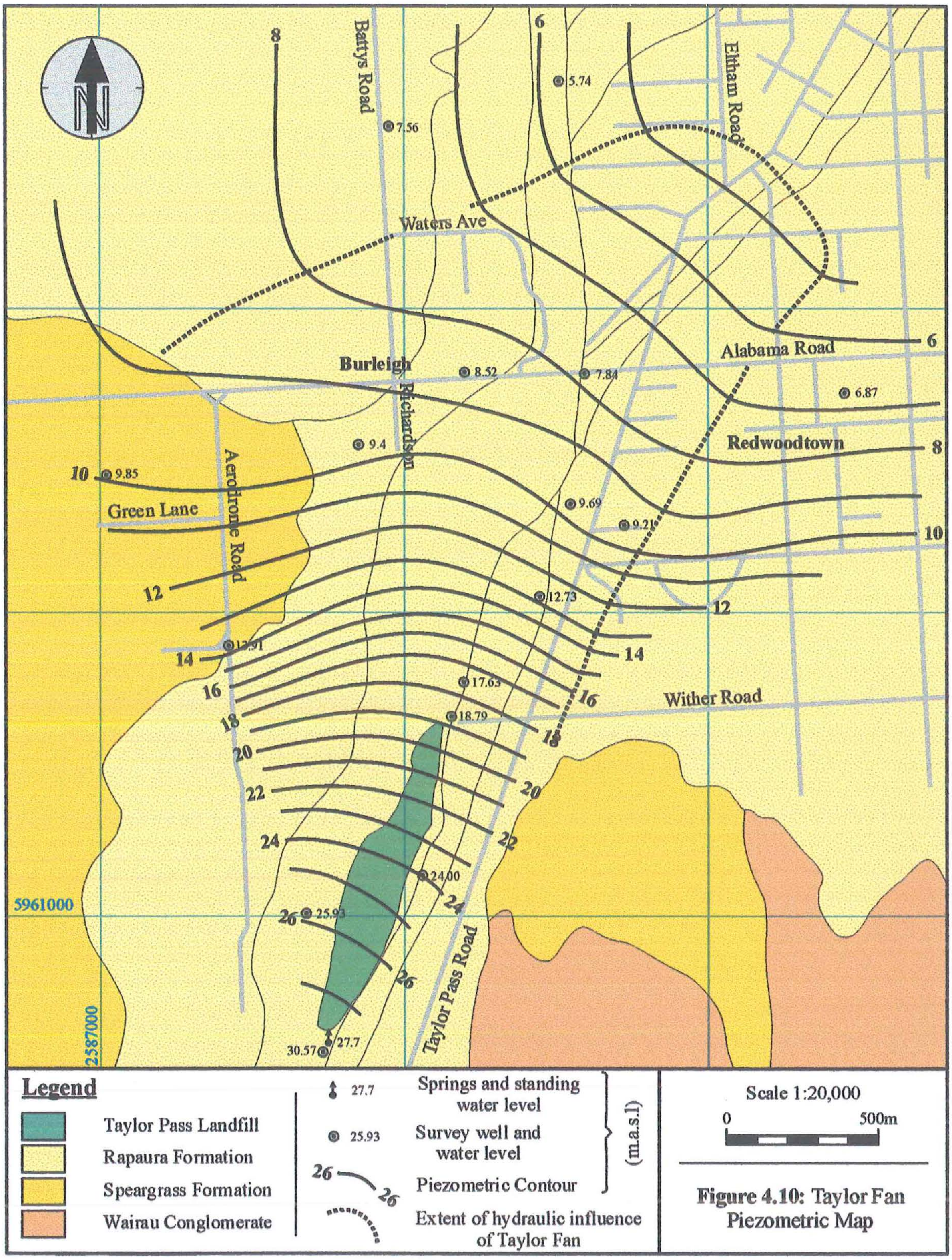


Figure 4.10: Taylor Fan Piezometric Map

Downstream of the Taylor Pass Landfill the lateral extent of the Rapaura gavels narrows, with Speargrass Formation gravels encroaching from the Wither Hills in the east and from the Speargrass lobe of the Taylor Fan in the west. Higher proportions of fine interstitial material in Speargrass Formation compared with the younger Rapaura gravels results in a decrease in hydraulic conductivity across the Rapaura-Speargrass boundary. Darcy's Law for flow through an aquifer states that: $Q = -KA\frac{dh}{dl}$ where, Q = flow (m^3/s)

K = hydraulic conductivity (m/s)

A = cross sectional area of flow (m^2)

dh/dl = hydraulic conductivity.

Although there is little control of the hydraulic gradient within the Speargrass Formation on the western margin of the valley, the effect of the decreased hydraulic conductivity due to fine interstitial material is to effectively decrease flow through the Speargrass Formation. Thus the Speargrass Formation effectively acts as a barrier to horizontal groundwater flow. Within the Rapaura Formation then the decrease in flow into the Speargrass Formation becomes the governing factor in the equation above, and the hydraulic gradient within the Rapaura Formation consequently increases near the Speargrass Boundary.

Steepening of the hydraulic gradient is likely also be a result of the influence of the more dominant Wairau Aquifer with a prevailing west to east flow direction. As groundwater flowing in a northerly direction through Rapaura gravels of the Taylor Fan reaches the boundary of the Wairau Aquifer, the dominant Wairau Aquifer waters restrict the northwards movement of Taylor Pass waters, which are incorporated into the main flow of the Wairau in the vicinity of New Renwick Road. The easing of the Taylor Pass hydraulic gradient from the south end of Richardson Road to Waters Ave reflects the gradual change between Taylor Pass dominated flow to Wairau dominated flow north of Waters Avenue.

Seasonal Variation in Groundwater Level

Variation in groundwater levels has been monitored in the upper Taylor Fan area at P28/W3387 approximately 50m south of the Taylor Pass Landfill over the period from April 1999 to March 2000 by means of an installed pressure transducer and data logger. Figure 4.10 shows the variation of water level over this period as being less than 0.5 m. The small fluctuation range in water level at P28/W3387, is reflected by the presence of constantly flowing springs located 20 m north of P28/W3387 and discharging into the diversion drain running around the southern and western perimeters of the Taylor Pass Landfill.

The correlation between rainfall and groundwater level for June 1999 (Figure 4.11) indicates that groundwater fluctuations generally respond to rainfall in the immediate area, with a lag response of approximately two days. Small oscillations evident at the end of June during the declining groundwater phase, however, are likely to be controlled by rainfall higher in the catchment. Monthly graphs of water level correlated with daily rainfall are given in Appendix 4.

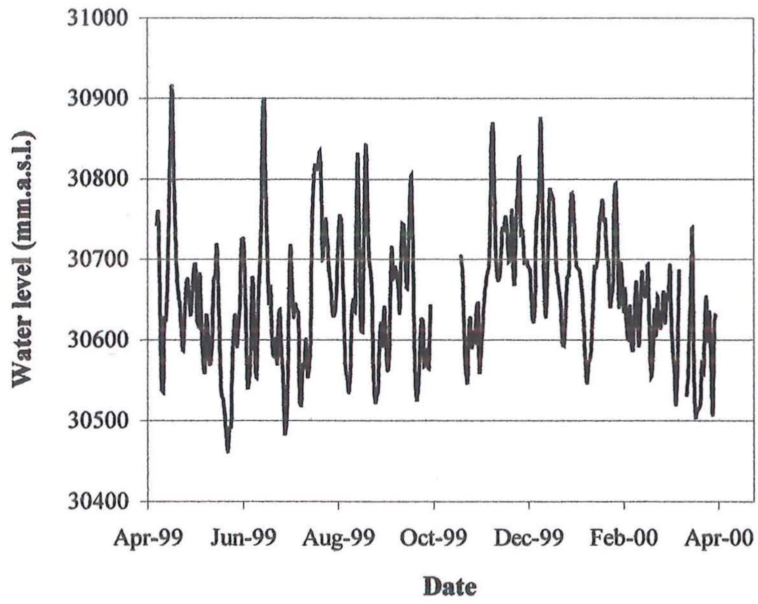


FIGURE 4.10: VARIATION IN WATER LEVEL AT P28/W3387 – APRIL 1999 TO APRIL 2000.

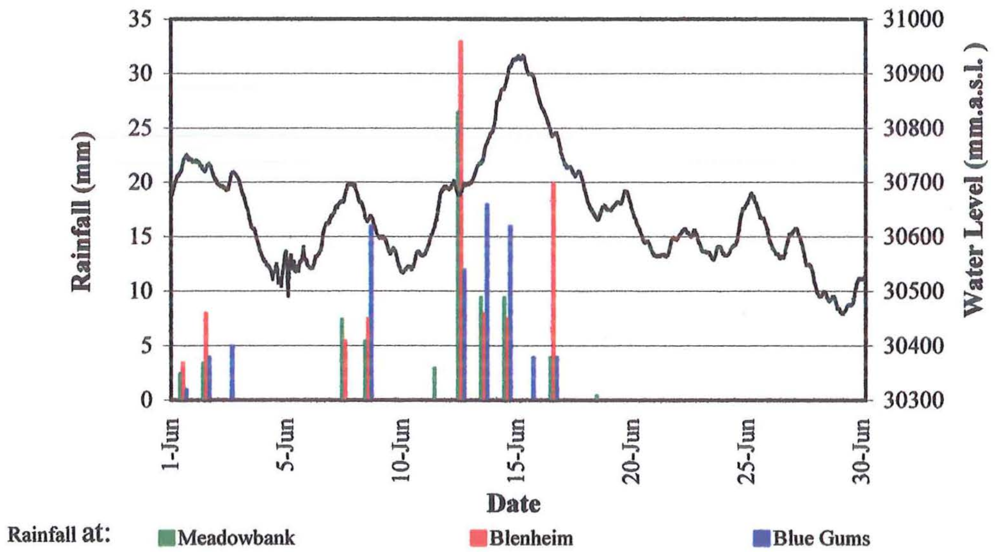


FIGURE 4.11: COMPARISON OF GROUNDWATER LEVEL (P28/W3387) AND RAINFALL FOR JUNE 1999.

Wells lower down the Taylor Fan surface show a greater groundwater fluctuation range than the upper Taylor Fan area described above. For example, P28/W2540 situated north of Brayshaw Park on New Renwick Road shows a seasonal variation of approximately 2.5 m (refer Appendix 4 for data), wells between New Renwick Road and the north end of the Taylor Pass Landfill show ranges of 1 – 1.7 m, and on the eastern side of the landfill P28/W3002 shows a seasonal variation of 0.5 m (similar to P28/W3387).

Local Groundwater Seepage Velocity

Because the tested wells do not penetrate the full extent of the aquifer, and in fact the depth of the tested aquifers is largely undetermined, calculations of seepage velocity (V_s) are based on effective depths of pumping wells as previously discussed in Section 4.4.3. Seepage velocity then has been calculated using the following relationship:

$$V_s = -K \times \frac{1}{n_e} \times \frac{dh}{dl}$$

where, K = Hydraulic conductivity $\left(= \frac{T}{d} \right)$

T = aquifer transmissivity (m^3/day/m), which is based on

d = effective depth of pumping/aquifer thickness (m) (refer Section 4.4.3),

n_e = effective porosity (dimensionless), and

dh/dl = hydraulic gradient.

The effective porosity of Speargrass and Rapaura gravels are a function of the proportions of fine material within the gravels, and are estimated at 25% and 30% respectively.

Results of investigations of hydraulic characteristics indicate that hydraulic conductivities in the mid to upper Taylor Fan area are likely to be 2 x 10^{-7} m/s to 6 x 10^{-4} m/s in sediments of the Speargrass Formation compared with 3 x 10^{-4} m/s to 8 x 10^{-4} m/s for sediments of the Rapaura Formation forming the modern river channel and recent flood plains. Taking into account the hydraulic gradient from the Taylor Fan piezometric surface as ranging between 1/180 and 1/25, calculated seepage velocities range from 4 x 10^{-4} m/day to 8.3 m/day through Speargrass Formation Gravels and 0.5 m/day to 9 m/day through the more permeable Rapaura Formation. The maximum seepage velocity calculated for the Speargrass Formation is likely to be overestimated as the hydraulic gradient does not appear to be as steep in the Speargrass Formation as in the adjacent Rapaura Formation. Likewise seepage velocities within areas of the Rapaura Formation dominated by overbank deposits may fall below predicted seepage velocities. This is emphasised by difficulty in pumping of monitoring wells to be discussed in Section 5.4.4.

4.5 Taylor Pass Landfill Hydrogeology and Water Balance Analysis

4.5.1 Introduction

The following section involves the investigation of hydrological and hydrogeological aspects of the Taylor Pass Landfill site. Moisture within and infiltrating through a landfill from meteoric, surface and groundwater sources is of critical importance to both the rate of decomposition of waste and the rate of leachate production. The overall water balance within a landfill is essential for establishing the proportion of that leachate which is derived from meteoric waters and that from infiltrating groundwaters in order to effectively minimise leachate production. The main components of the landfill water budget were introduced in Section 2.3 (Figure 2.2). The moisture input components applicable to the Taylor Pass Landfill are precipitation, liquid waste disposal, and groundwater. Surface run-on is absent and surface seepage is negligible.

A number of methods of water flux calculations have been devised in order to ascertain the rate of water input to a landfill. The Water Balance Method (Thornthwaite and Mather, 1957) applied to solid waste disposal sites (Fenn *et al.*, 1975) forms the basis of water flux methods and predicts leachate generation by calculating moisture availability and transfer in cover soils. The principal source of moisture is precipitation and thus it is primarily a method for calculating only the surface water component. The Water Balance Method does not incorporate the input of water to the landfill by groundwater movement. Standing water noted at the base of the former gravel extraction pit of the Taylor Pass Landfill site (pers. com. J Dovey, 1999), suggests that the landfill site does intercept the groundwater table, and thus that leachate generation by groundwater infiltration is certainly significant and must also be considered.

The Hydrologic Evaluation of Landfill Performance (HELP) model (Schroeder *et al.*, 1994) allows for modelling of multi-component engineered cover layers (refer Section 2.3), and accounts for water movement through the landfill itself. Calculation by the HELP method is considered excessive and hence inappropriate for the assessment of water infiltration at the Taylor Pass Landfill based on the irregular depth, and nature and spatial variation of the cover materials. The Water Balance Method (Fenn *et al.*, 1975) thus forms the basis of the following surface water budget calculations. Groundwater infiltration is also assessed based on local hydraulic characteristics and piezometric data discussed in the previous section, and site characteristics discussed in Section 3.6.

4.5.2 Surface Water Balance Method

The Water Balance Method was developed in the soil and water conservation field by Thornthwaite and Mather (1955), and applied to sanitary landfills by Fenn et.al. (1975). Based on the interactions between precipitation, evapotranspiration, surface runoff and soil moisture storage, the method relates site-specific climatological and geotechnical information to predict the rate of leachate generation by meteoric waters. Precipitation will either run off the landfill surface, evaporate directly back into the atmosphere from the soil surface (and to a lesser extent from vegetation surfaces), be utilised by plants through transpiration, or increase the moisture content of the cover-soil to field capacity. Above field capacity, moisture within the cover-soil will percolate downwards under the force of gravity to form leachate.

The surface-water balance for the Taylor Pass Landfill has been calculated on a daily basis over a twelve-month period from January 1999 to December 1999. A discussion of the main water flux components follows.

Precipitation

Average monthly rainfall data from 5 stations in the Upper and Lower Taylor catchment are shown in Table 4.5. All stations are maintained by the MDC with the exception of the Blue Gums site, which is monitored by Blue Gums Regional Landfill personnel. Tinpot (Grid ref. 8300/5150) and Beneagle (Grid ref. 9200/5700) Stations indicate the difference in precipitation in the upper catchment area in comparison to the Lower Taylor Fan Stations. Upper catchment areas receive approximately 1.5-2 times the annual rainfall of the lower catchment stations.

Meadowbank, Blue Gums and Blenheim Stations are all located in the Lower Taylor catchment to the southwest, southeast and north of the Taylor Pass Landfill respectively. The averaged daily precipitation from Meadowbank, Blue Gums and Blenheim stations is assumed to be representative of the rainfall received at the Taylor Pass Landfill based on its geographical location with respect to the monitoring stations. Appendix 4 lists the average daily precipitation from 1 January 1999 to 31 December 1999 extrapolated by the author from Meadowbank, Blue Gums and Blenheim stations.

Rainfall intensity data from Rae (1987) for the Blenheim rain gauge is shown graphically in Figure 4.12. Table 4.6 shows maximum rainfall depth-duration values from extrapolated rainfall data for the Taylor Pass Landfill in 1999. The 24-hour and 48-hour maximum intensities of the Blenheim data and extrapolated 1999 landfill data are comparable, with the computed 72-hour maximum intensity being slightly elevated above real data from the Blenheim station. These data do however indicate that the daily rainfall used for the calculation of the landfill water budget is representative of "typical" annual rainfall in the area.

	Tinpot	Beneagle	Meadowbank	Blue Gums	Blenheim
Jan	73	55	37	20	49
Feb	59	52	29	41	45
Mar	68	67	43	42	52
Apr	57	54	39	33	52
May	73	57	34	33	53
Jun	92	70	66	61	64
Jul	140	98	98	109	74
Aug	96	85	35	40	64
Sep	105	65	41	38	52
Oct	73	64	47	61	57
Nov	84	69	52	54	54
Dec	63	60	32	52	58
Total	982	798	552	583	674

TABLE 4.5: AVERAGE MONTHLY RAINFALL – TAYLOR PASS AREA.

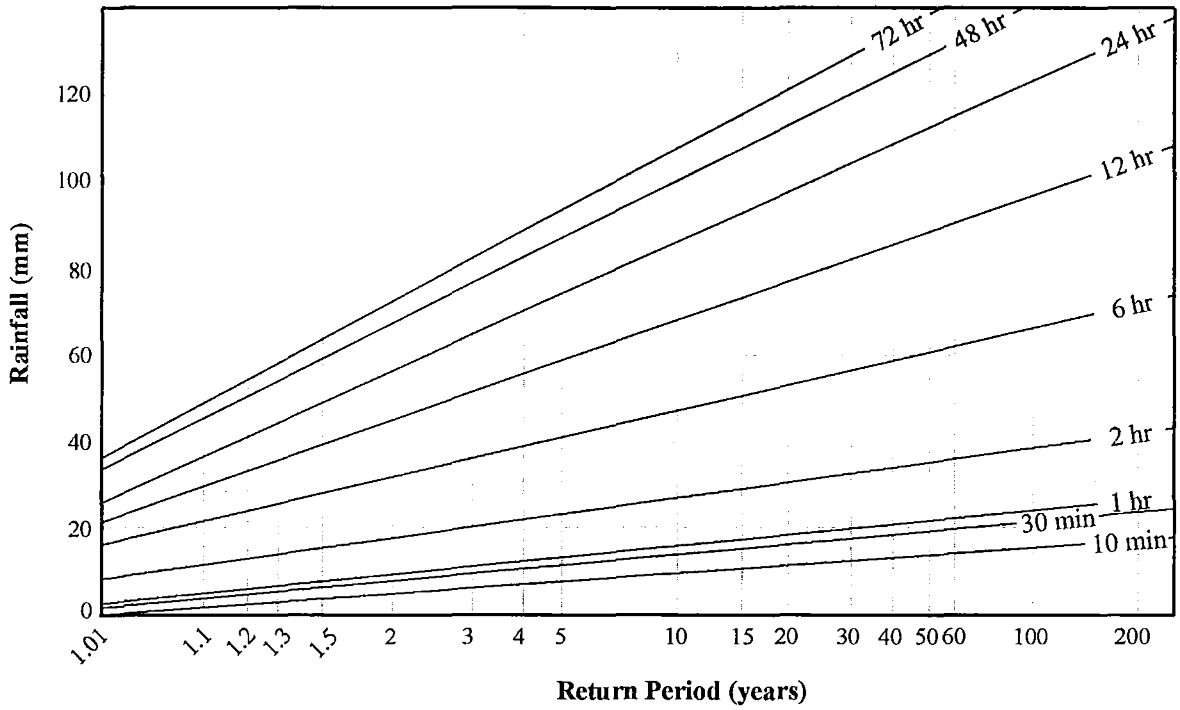


FIGURE 4.13: RAINFALL DEPTH – DURATION – FREQUENCY CURVES; BLENHEIM (FROM RAE, 1987).

Total annual rainfall (mm)	574
Maximum 24 hour rainfall (mm)	24
Maximum 48 hour rainfall (mm)	37
Maximum 72 hour rainfall (mm)	50

TABLE 4.6: RAINFALL INTENSITY ANALYSIS FOR EXTRAPOLATED TAYLOR PASS LANDFILL RAINFALL DATA – 1999.

High intensity rainfall events with return period greater than 1 year are likely to have an effect on volumes of infiltration and percolation through the landfill surface, however long duration events that may lead to saturation of the soil cover and hence increased downwards percolation are expected to be more significant. This aspect of the landfill water budget is further discussed with respect to the soil moisture storage capacity in following sections.

Surface Run-off

Where no on-site gauging of runoff exists, surface runoff is commonly calculated by either the rational method (Chow, 1964) or the curve number method (US Department of Agriculture – Soil Conservation Service (US DoA SCS), 1972).

The rational method is expressed as:

$$R = C \times W_p$$

where, R = surface run-off peak discharge (mm),

C = runoff coefficient, and

W_P = precipitation (mm)

Runoff in this method is considered to be a direct proportion of the total rainfall and takes no account of the effects of antecedent rain on the ability for soils to absorb moisture. Numerous methods for the estimation of the run-off coefficient have been proposed based on factors of vegetation cover, soil type and slope angle (e.g. American Society of Civil Engineers, 1960; Perry, 1976; Salvato *et al.* 1971. all in Lu *et al.*, 1985).

At the Taylor Pass Landfill it was expected that due to the dry climate and infrequent high intensity rainfall events, the condition of the soil with respect to antecedent rainfall would be a major controlling factor in the assessment of run-off. For this reason the US DoA SCS (1972) method of curve fitting has been chosen as a the most valid estimate of runoff (Figure 4.13). A basic description of the method and calculation of the curve number applied to the Taylor Pass Landfill can be found in Appendix 4.

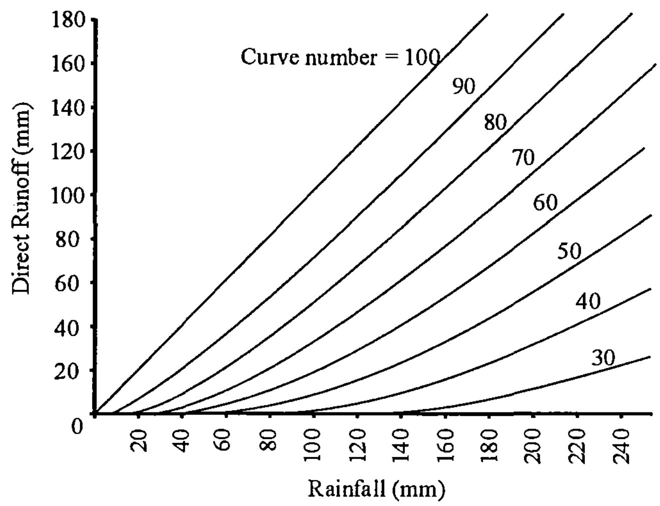


FIGURE 4.14: GRAPHICAL REPRESENTATION OF THE ESTIMATION OF RUNOFF BY THE CURVE NUMBER METHOD (AFTER US DOA SCS, 1972). NB. THE METHOD FOR CALCULATING THE CURVE NUMBER IS DESCRIBED IN APPENDIX 4.

Assumptions applied to the curve number calculation based on site-specific characteristics of the Taylor Pass Landfill, and the effects of assumptions on the estimated runoff, are as follows:

- The landfill is considered to have no dominant growing or dormant season and the Antecedent Moisture Condition Class (AMC) used is based on dormant season ratings where consumptive use by vegetation is a minimum. This conservative assumption effectively reduces the resultant curve number, thus increasing the amount of rainfall required to overcome the initial phase of 100 % infiltration (refer Figure 4.13).

- Curve numbers for specific hydrologic soil group, land-use and cover, and hydrologic conditions present over the landfill surface contribute proportionally to the overall curve number used in the final spreadsheet calculations for runoff (refer Appendix 4). Actual runoff therefore may be over or under-estimated for certain areas of the landfill due to averaging of defining parameters.

Infiltration

Field infiltration testing was attempted using a double ring infiltrometer in order to calibrate calculated and actual infiltration rates, and to define the relationship between calculated infiltration rates and laboratory tested cover soil permeabilities.

Desiccation crusting of clay-rich cover soils occurs at the Taylor Pass Landfill due to extreme drying in the summer season. Testing was attempted in March 2000 immediately after the dry season, at which time it was not possible to penetrate sufficiently through the surface layers to isolate a cohesive body of soil without destroying the soil structure and equipment. Diminishing the integrity of the soil body to be tested in effect creates new infiltration paths, drastically increasing the soil's vertical infiltration capacity and thus masking the true infiltration capacity of the soil. Field infiltrometer testing thus proved to be an ineffective method due to the dry nature of the soils. Infiltration testing is likely to be more effective if carried out following the wet season, prior to desiccation of the upper soil layers, and is recommended for further investigations.

Infiltration for the Taylor Pass Landfill water balance analysis has been calculated wholly on the following relationship:

$$Infiltration = Precipitation - Runoff$$

where infiltration includes all precipitation that is intercepted by vegetation as well as that which infiltrates the soil surface. Figure 4.14 shows the effect of surface vegetation on the rates of infiltration by means of both interception and transpiration as measured in test lysimeters.

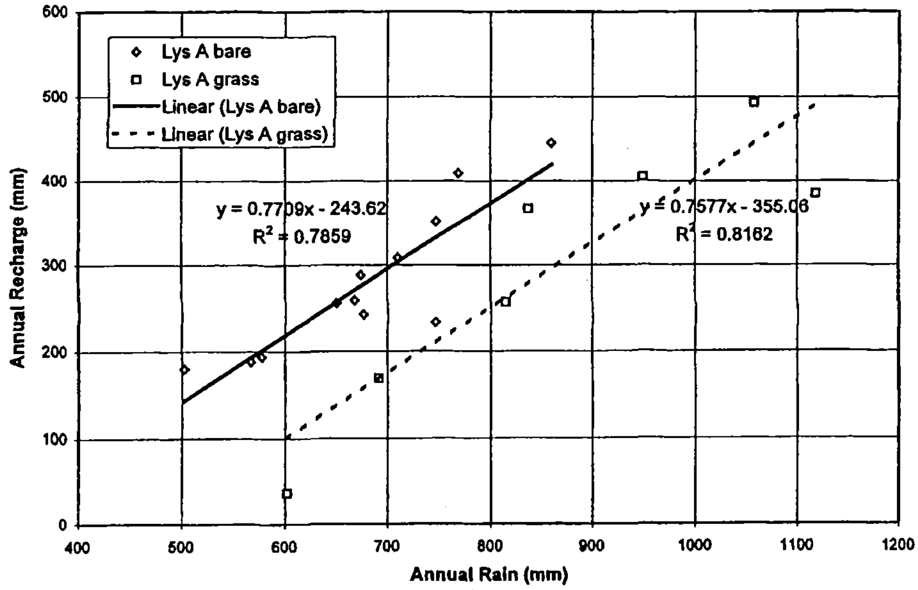


FIGURE 4.15: COMPARISON OF INFILTRATION THROUGH BARE AND GRASSED SOILS. (FROM THORPE AND SCOTT, 1999).

Evapotranspiration

Evapotranspiration is defined as the combined losses of water to the atmosphere by evaporation from the ground surface and through consumptive use by vegetation (transpiration), and is a function of both climatic and site conditions. Where transpiration is the exchange of water from the soil to the atmosphere through plant stomata, Hendricks and Hansen (1962, in Linsley *et al.*, 1972) report that rates of transpiration are approximately equivalent to rates of evaporation of water directly from leaves. If water availability to plants is not restricted, evapotranspiration can be thus assumed to be equivalent to the rate of evaporation from a free water surface. Kohler (1952, in Linsley *et al.*, 1972) based on a number of cases, suggests that free water evaporation rate and hence potential evapotranspiration from a vegetated soil surface (Linsley *et al.*, 1972) can be estimated from pan evaporation values by applying a pan coefficient, which has an average value of 0.7 yet ranges from 0.67 to 0.81. This approach is applicable only where the water availability for plants is not restricted, and the method gives no consideration to the variable transpiration rates of different vegetation types and densities.

The method has, however, been selected due to the availability of reliable pan evaporation data collected in the Wither Hills and Vernon Lagoon at the southern end of the Cloudy Bay coast. To account both for the restricted availability of water and for areas on the landfill where vegetation density is minimal, and to maintain a conservative estimate of evapotranspiration, a pan coefficient of 0.67 has been used for calculations in Appendix 4.

Pan evaporation data collected on site over 10 years at Vernon Lagoon, and over an unknown period of time at Wither Hills, show relatively similar trends, with Vernon Lagoon showing slightly higher rates through September and October (Figure 4.16). Differences in evaporation cannot be adequately explained but may be attributable to a coastal wind pattern or other microclimatological differences. For the purpose of water budget calculations, Wither Hills pan evaporation data has been used due to its close proximity to the Taylor Pass Landfill.

Transpiration rates vary depending on root depth and stomata. For grass and weed cover comparable to a significant proportion of the Taylor Pass Landfill cover, the typical consumptive use of water is approximately 1.8m/yr based on data from Lutton *et al.* (1979). Transpiration however, occurs only when soil water is available; due to the dry climatic nature and long periods without rainfall in the Taylor Pass Area, consumptive use will obviously be significantly lower than theoretical values.

For calculation purposes, it is assumed that:

- potential evapotranspiration will be equal to 67% of the measured monthly average pan evaporation;

- any infiltration (precipitation less runoff) in excess of potential evaporation will contribute directly to soil moisture storage; and
- if potential evapotranspiration exceeds precipitation, available soil moisture will be depleted correspondingly (see following section).

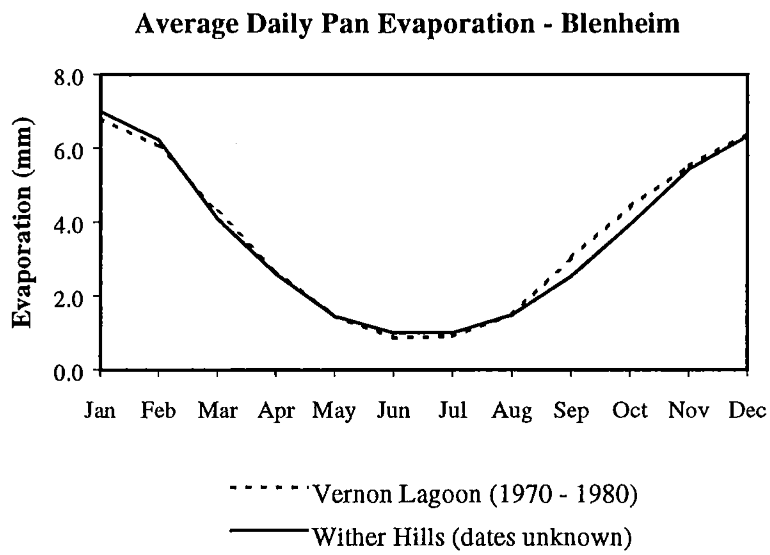


FIGURE 4.16: PAN EVAPORATION DATA FROM TWO BLENHEIM SITES (DATA FROM MDC).

Soil Moisture Storage

Moisture is stored in soil as hygroscopic and available water. Hygroscopic water is that which is tightly bound to the soil particles and cannot be removed from the soil by plants. Available water is that which is retained within the interstices of the soil yet is unable to migrate vertically under gravitational forces but is available to plants and/or the atmosphere for transpiration and evaporation processes respectively (Fenn, 1975). Soil moisture storage capacity depends on both the nature and structure of soils, and the depth of cover soil and the root depth of vegetal cover. Figure 4.17 illustrates the relationship between wilting point, field capacity and saturation of different soil types.

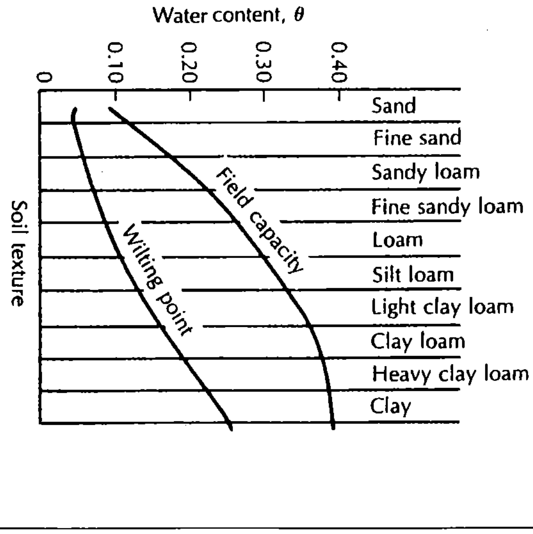


FIGURE 4.17: WATER HOLDING CAPACITY OF SOILS (US DEPT. OF AGRICULTURE 1955, IN FETTER, 1994).

In order to assess the field capacity of landfill cover soils a selection of saturated permeability samples (TPSS 7, 8, 9, 10 and 11, refer Appendix 3) were allowed to free drain under the influence of gravity for 65 hours. Permeability test samples were adjusted by removal of large grain sizes in order to meet standard permeability test methods. The proportion of oversized material is also indicated in Table 4.7.

Based on drainage data and taking into account the effect of the removal of coarse sediments, the average field capacity for the Taylor Pass Landfill for the purpose of surface water budget calculations only has been estimated at approximately 20-22% moisture content or equivalent to that of a "sandy loam" (Figure 4.16). In the absence of a more accurate measure of average field capacity over the landfill this is considered to be a reasonable assumption as those samples tested are located within the uncapped and un-landscaped section of the landfill (refer Section 3.6.2, and Appendix 3). Both the capped area of the landfill and the landscaped area surrounding the Transfer Station and Recycling Centre are expected to have a higher field capacity, and thus over the landfill as a whole are likely to compensate for any overestimation of field capacity based on tested samples.

Sample Number	**Removed material >4mm** (% by weight)	**Moisture content after 65 hours (%)**
TPSS-7	40	36
TPSS-8	22	30
TPSS-9	44	20
TPSS-10	37	22
TPSS-11	45	22

TABLE 4.7: MOISTURE CONTENT OF ADJUSTED LANDFILL COVER SOIL SAMPLES FOLLOWING 65 HOURS FREE DRAINING UNDER THE INFLUENCE OF GRAVITY.

Given the largely undefined yet probably variable depth of cover soils, and the variable vegetation over the Taylor Pass Landfill (refer Chapter 3), the average soil moisture storage capacity has been applied over a nominal depth of 0.7m, giving a soil cover moisture storage capacity of 120-130mm. The soil moisture retained after different amounts of potential evapotranspiration then is calculated using tables provided by Thornthwaite and Mather (1957) for soils with soil moisture capacity of 125mm.

Results

Data has been analysed on a daily basis for the Taylor Pass Landfill in order to more accurately assess the volume of precipitation percolating through the landfill cover. Full results of analysis are given in spreadsheet format in Appendix 4.

Percolation into the Landfill will only occur when the soil moisture capacity is exceeded, and infiltration exceeds the potential evapotranspiration. Results indicate that percolation into the

Taylor Pass Landfill based on parameters discussed above occurs only during July and early August, contributing 66 mm of water to the refuse body. Over an area of 23.8 ha, this corresponds to a total volume of 15700m^3 of infiltrating water.

Since groundwater is present in the base of the Taylor Pass Landfill, it is assumed that leachate generation would have begun immediately following refuse disposal, and hence calculations for the appearance of leachate based on the water balance equation have not been carried out.

Implications of Landfill Cover Integrity

Cracking and desiccation of clay rich cover soils at the Taylor Pass Landfill were noted during the summer season. Thorpe and Scott (1999) report shrinkage of soils in lysimeter tests over dry summer periods leading to the formation of gaps and preferential conduits for infiltration of precipitation around the internal perimeter of the lysimeter. The gaps are noted to close following precipitation events due to the swelling of clays within the soil. Thorpe and Scott (1999) suggest that the desiccation and cracking of soils is likely to influence infiltration rates during high "heavy summer rains".

The same theory has been adopted to the Taylor Pass Landfill, where high intensity summer rainfall events are rare. The influence of desiccation and cracking is thus considered negligible with respect to the long-term annual water budget, but it must be emphasised that the integrity of the cover layer will be of critical importance during rare high intensity rainfall events where precipitation is supplied to the landfill surface faster than the rate of infiltration into (and hence the swelling rate of) desiccated clay soils. Significant amounts of rainfall may in this case percolate through the cover layer before the moisture capacity of the cover soils reach field capacity.

4.5.3 Groundwater Infiltration

Background

Extraction of gravel at the Taylor Pass Landfill occurred to depths of between three and 5 m below the existing ground level, with the base of the pit generally coinciding with the groundwater table at the time (pers. comm. Jim Dovey, 1999). Aerial photographs indicate that standing water was present in the base of the pit following winter months, but none was evident following the drier summer months. It is likely then that during the winter months at least, groundwater infiltration through the base of the landfill may be adding to the rate of leachate generation. The measured fluctuation of groundwater in P28/W3387 (Section 4.4.4.) of 0.5 m suggests that even if water is present in the base of the landfill in winter, then during the dry summer months, the groundwater level is likely to be within 0.5 m of the base of the landfill provided the water level fluctuation range beneath the landfill is the same as in P28/W3387. The comparison of water level fluctuation of the measured well and the base of the landfill is considered a reasonable assumption.

In order to evaluate the volume of groundwater infiltrating through the Taylor Pass Landfill then, it is necessary to have an understanding of local groundwater flow, piezometric surfaces and the seasonal fluctuation in groundwater level. These aspects have all been covered with respect to the Taylor Fan in Section 4.3, and are here applied specifically to the Taylor Pass Landfill. The leachate collection system in use at the landfill is introduced and pumped leachate volumes are assessed.

Site Hydrogeology

Figures 4.18 and 4.19 shows cross sections constructed through the Taylor Pass Landfill based on trench logs (Appendix 2) and water level data from wells surrounding the landfill (Table 4.8).

Well Number	Location	Piezometric level as at 10 June 1999 (m.a.s.l)
P28/W2539	40 m NE of TPL*	18.55
P28/W3002	25 m E of TPL	24.00
P28/W3387	90 m S of TPL	30.57
P28/W3388	100 m W of TPL	25.93
Standing water level beneath springs – 03/12/99 (m)		27.7

TABLE 4.8: PIEZOMETRIC CONTROLS ON THE PHREATIC SURFACE WITHIN THE TAYLOR PASS LANDFILL. NB: THE STANDING LEVEL OF WATER BENEATH THE SPRINGS AT THE SOUTH END OF THE LANDFILL IS ASSUMED TO VARY IN THE SAME MANNER AS P28/W3387.

The piezometric surface in Figures 4.18 and 4.19 are extrapolated from Figure 4.10 and suggest that groundwater intercepts the landfill only in the southern portion of the landfill (Cross section A-A*, Figure 4.19). The groundwater level does, however, remain within 1.5 m of the base of the landfill along Cross section L-L* through the southern and mid sections of the landfill, decreasing below this level at the northern end of the landfill where the hydraulic gradient steepens. It is also possible that some mounding of the phreatic surface may occur beneath the landfill, although this cannot be modelled because of the absence of wells through the landfill. Seepage on the eastern margin of the landfill above the projected water table indicates that some perching of water bodies occurs within the Rapaura gravels and overbank deposits that are expected to periodically flush the base of the landfill.

Possible scenarios that may account for the presence of water at the base of the landfill prior to filling and above the projected water table in Figure 4.19 include:

1. Perching of the water table above the "blue pug" layer at the base of the landfill with infiltration occurring from the south (Figure 4.20a).

2. Complex interfingering of a number of overbank deposit layers, beneath the known "blue pug" layer, hosting a perched water body, which may appear as springs at the base of the landfill (Figure 4.20b).

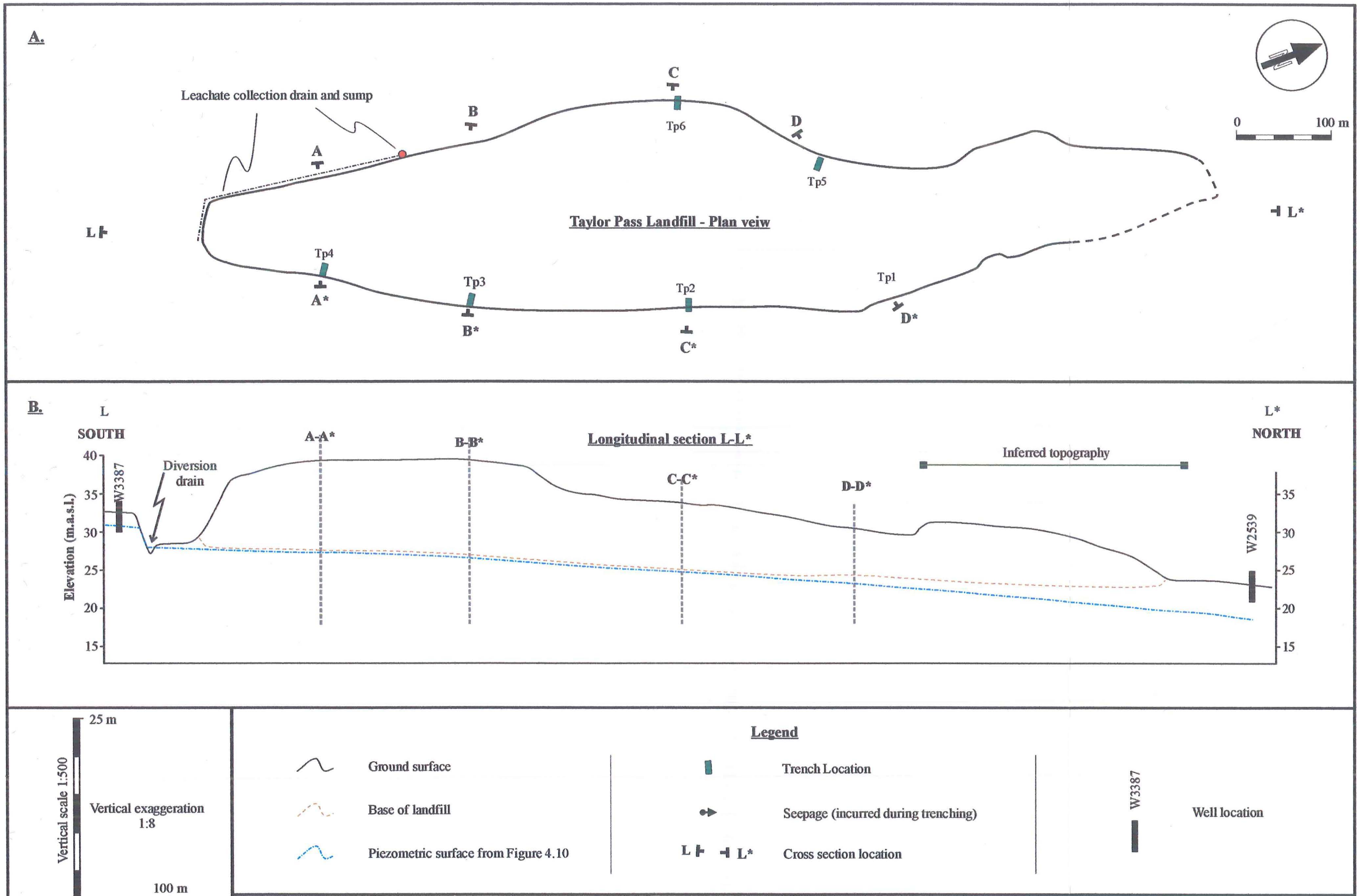


FIGURE 4.18: A) LOCATION OF TRENCHES AND LONGITUDINAL AND LATERAL CROSS SECTIONS (FIGURE 4.19), B) LONGITUDINAL CROSS SECTION THROUGH THE TAYLOR PASS LANDFILL.

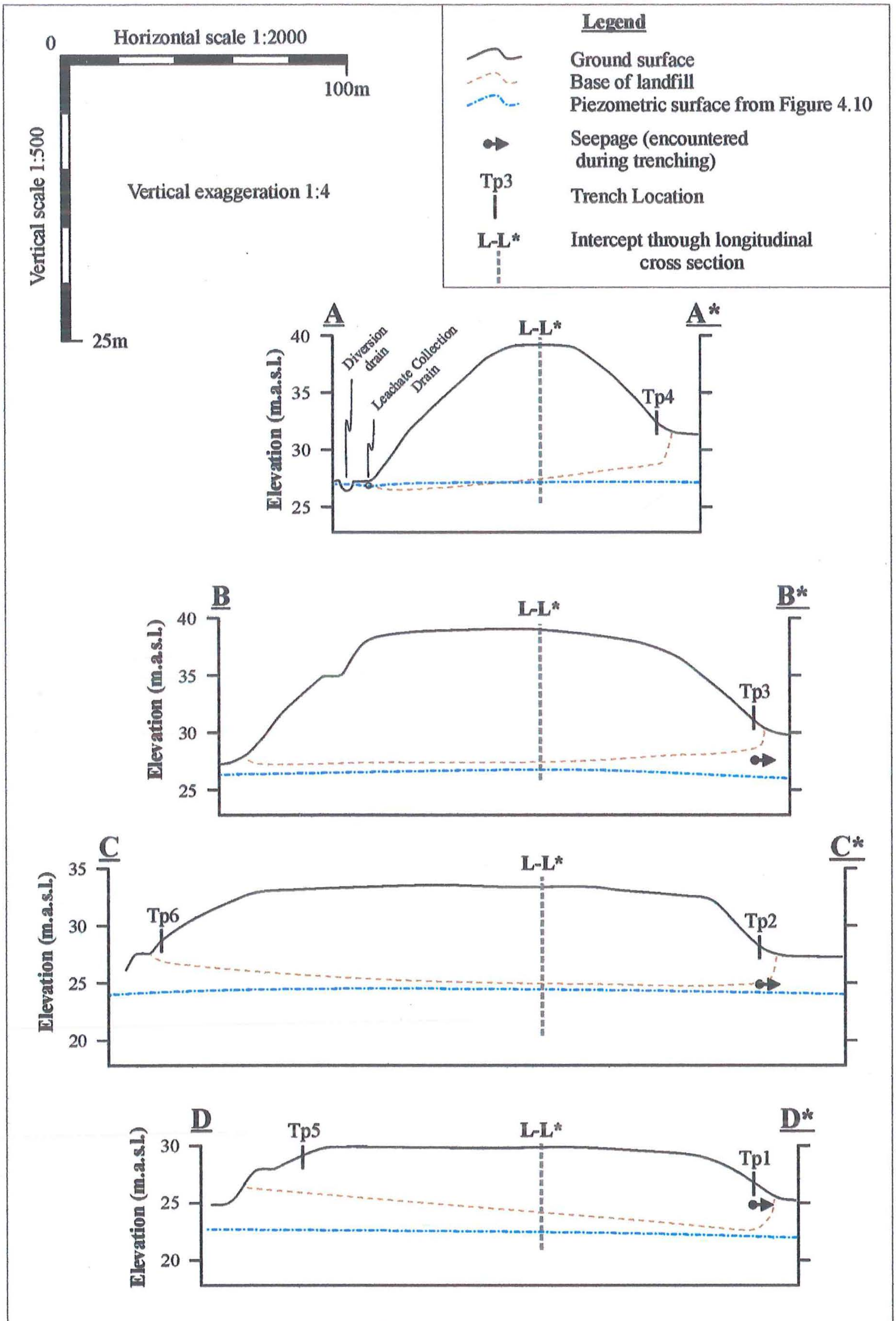


FIGURE 4.19: LATERAL CROSS SECTIONS THROUGH TAYLOR PASS LANDFILL. REFER FIGURE 4.18 FOR LOCATIONS. THE DEPICTED PIEZOMETRIC SURFACE IS EXTRAPOLATED THE LOCAL PIEZOMETRIC MAP (FIGURE 4.10) AND DOES NOT ACCOUNT FOR ANY POSSIBLE MOUNDING OF GROUNDWATER BENEATH THE

3. The introduction of groundwaters from the east via perched and/or unconfined water bodies in the older Rapaura Formation gravels (rg and ra of Brown, 1981) towards the modern flood plain gravels (fa) (Figure 4.18c and d).

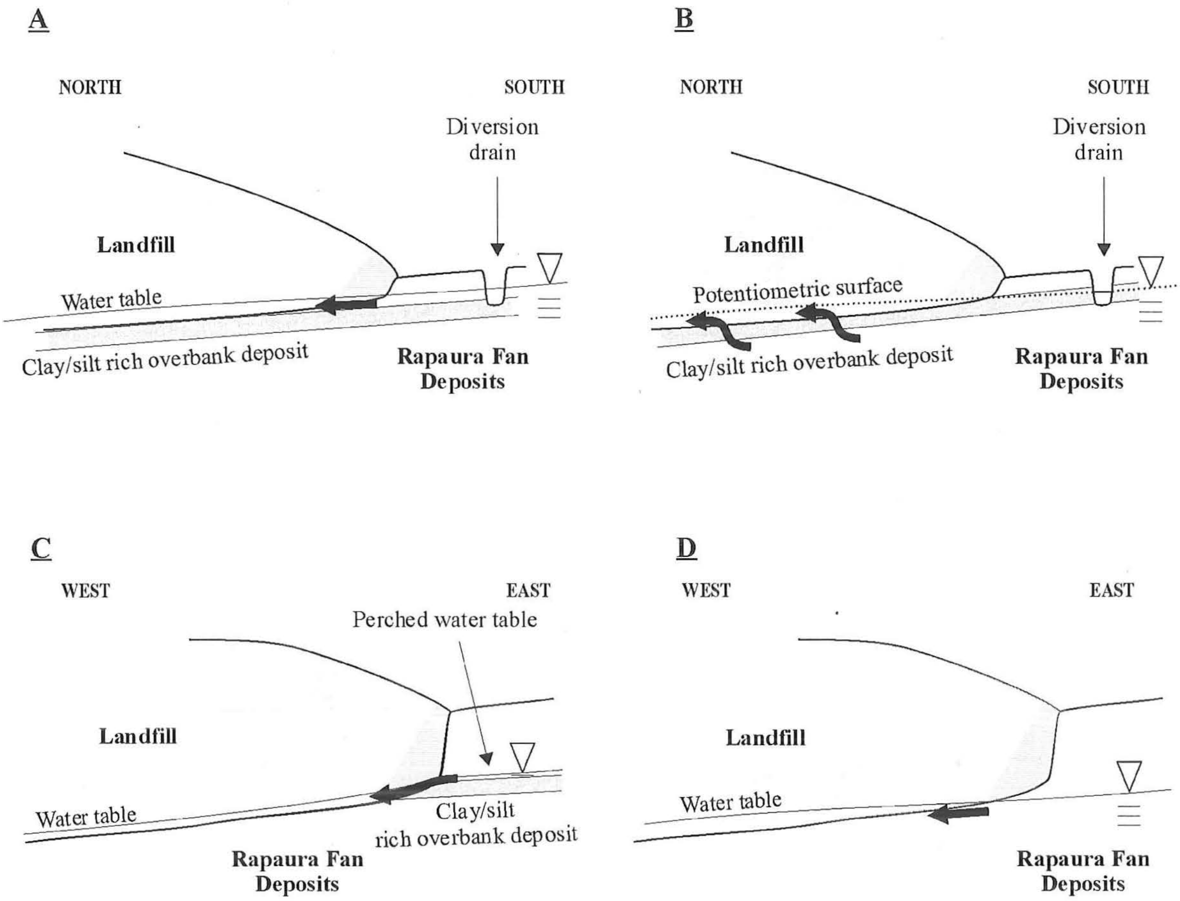


FIGURE 4.20: POSSIBLE MECHANISMS OF GROUNDWATER FLOW INTO THE TAYLOR PASS LANDFILL. REFER TO TEXT FOR DISCUSSION.

Given the presence of seepage on the eastern margin of the landfill in trenches 1, 2 and 3 (Appendix 2), and the location of springs to the south of the landfill at an elevation of approximately 41 m.a.s.l., it appears likely that the third scenario may contributing to groundwater within the landfill. If this is the case then groundwater entering the landfill from the east will almost certainly remain perched above the "blue pug" layer as indicated in scenario 1. The presence of springs at the base of the landfill indicate that contribution of groundwater by scenario 2 is also likely, yet no obvious deeper confining layer is noted in P28/W3387 or P28/W3002.

The constant flow of springs at the southern end of the landfill, and the relatively small piezometric range of P28/W3387, indicates that the groundwater level through the south end of the landfill is likely to be relatively stable. P28/W2539 displays a more variable piezometric range of approximately 1.5 m, indicating that the groundwater level within the landfill is likely controlled by backing up of groundwater from a northern direction rather than by surges in supply from the south. Within the landfill, then, the piezometric range will be greater in the older northern portion of the landfill where there is no control over the movement of leachate.

Piezometers were not installed in the landfill body as a part of this project due to both the risks associated with drilling into gas producing landfill body, and the possible risk of penetrating through the "blue pug" layer and forming an additional conduit for vertical migration of leachate. Quantification of the level to which groundwater will rise due to any mounding is impossible in the absence of monitoring bores within the landfill, thus the assessment of groundwater infiltration into the landfill can remain only speculative. In retrospect, it is deemed essential to overcome these difficulties and to insert at least three piezometers down the length of the landfill to accurately ascertain the groundwater level (and fluctuation range) within the landfill.

Leachate Collection System and Pumping Volumes

Leachate recirculation is the current method of leachate disposal at the Taylor Pass Landfill. Simply, leachate is intercepted prior to discharge into the environment by a series of drainage systems and recirculated through the body of the landfill. By this process, leachate is effectively reduced in volume and altered in strength primarily by mechanisms of adsorption and filtration. Aerobic and anaerobic decomposition is also enhanced, promoting waste stabilisation.

In early 1996, a leachate collection system was installed around the southwestern and southern perimeter of the Taylor Pass Landfill. Initial plans for the system consisted of a bund keyed into low permeability blue clay and silts known to exist under the southern and western margins of the landfill. A collection pipe was to be installed on the landfill side of the bund, diverting collected leachate into a settlement/oxidation pond. Further recirculation of leachate back into the landfill through soakage drains was proposed prior to discharge of leachate to the western drain via a constructed wetland. Neither the wetland nor the bund has been constructed (Connell Wagner Ltd, 1998).

The current collection system consists of a 100 mm diameter perforated subsoil drain feeding into a collection sump. Collected leachate is pumped to two filter beds on top of the landfill (Figure 4.21), and each consists of a 20m length of 100 mm perforated land drain set in a layer of drainage metal. The filter beds are capped with low permeability clay to prevent mixing with meteoric waters. An overflow sump constructed at the end of the filter bed collects any excess leachate pumped into the filter bed. Recorded machine hours and switch counters on the collection sump at the end of the perimeter drain allow the calculation of the volume of leachate passing the through the leachate collection system.

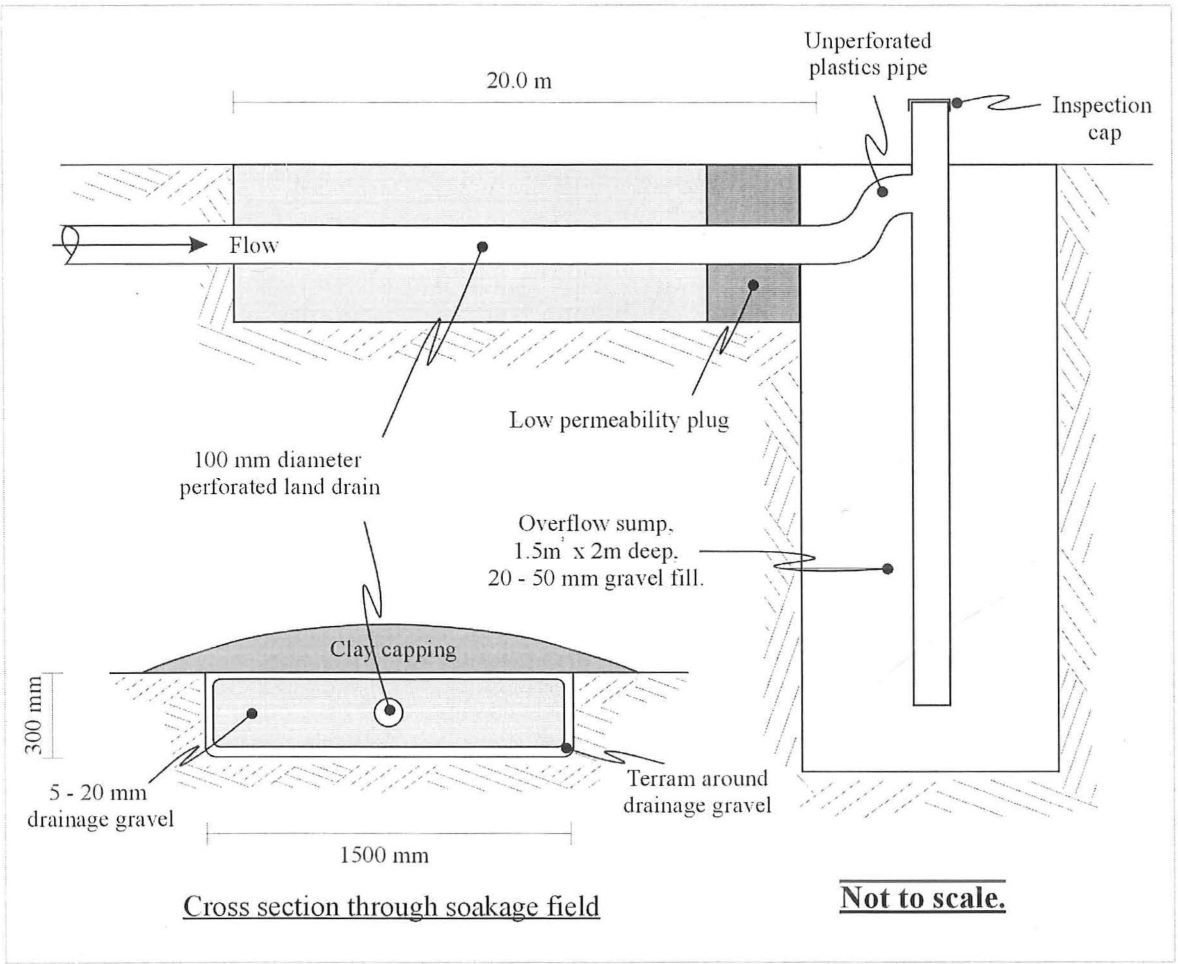


FIGURE 4.21: LEACHATE SOAKAGE FIELDS, TAYLOR PASS LANDFILL, BLENHEIM (AFTER ROYDS CONSULTING LTD, 1996).

The filter beds are located towards the northern end of the landfill site, down gradient of the northern end of the collection drains. Because the landfill is unlined, the effectiveness of the leachate collection system with respect to leachate treatment capabilities is significantly reduced. The system effectively prevents raw leachate from entering the diversion drain around the perimeter of the landfill, but when reintroduced to the body of the landfill it is free to pass immediately into through-flowing groundwater. The collection system thus works as a leachate diversion rather than as a treatment or containment system.

Given the location of the collection drains between the perimeter of the landfill and the diversion drain, it appears likely that any uncontaminated groundwater seeping from the diversion drain will also be intercepted and collected through the leachate collection drains. The proportion of intercepted leachate to uncontaminated groundwater cannot be determined without accurate investigations of the water table within the landfill. It is postulated, however, that given the likely effects of groundwater mounding within the landfill and the probable contrast in hydraulic conductivity between the refuse body and the "blue pug" layer expected to exist between the diversion drain and leachate collection drain, uncontaminated groundwater is likely to constitute only a small portion of the total volume of liquid passing through the system. In calculation of groundwater-generated leachate in the next section, uncontaminated groundwater entering the leachate collection system is disregarded.

Table 4.9 lists the pumping hours and corresponding pumped volumes over the period from 6 November 1998 to 12 November 1999, based on a 35 l/s-pumping rate. It must be emphasised that the pumped volumes do not represent the total amount of leachate produced at the Taylor Pass Landfill from groundwater, surface water or combined sources as the collection drain only runs southwards from the leachate sump (Figure 4.18). Insufficient data on water levels within the landfill exist to accurately calculate the total volume of leachate produced within the landfill, however, from figures 4.18 and 4.19, it appears that the area of the landfill which is drained by the leachate collection system is the only area of the landfill which is permanently intercepted by the groundwater table. A more accurate assessment of the area and the fluctuation of the area intercepted by groundwater would be possible with the installation of piezometers within the landfill body.

Date	Pumping Hours	Volume Pumped (m^3)
6 Nov 1998 – 4 Dec 1998	22.71	2861
– 8 Jan 1999	0	0
– 5 Feb 1999	8.2	1033
– 5 Mar 1999	9.43	1188
– 1 Apr 1999	9.75	1228
– 2(?) May 1999	8.94	1126
– 5(?) June 1999	Main switched off	-
– 2 July 1999	13.47	1697
– 6 Aug 1999	50.05	6306
– 20 Aug 1999	18.03	2272
– 3 Sep 1999	13.96	1759
– 1 Oct 1999	17.31	2181
– 12 Nov 1999	18.92	2384

TABLE 4.9: DETAILS OF LEACHATE PUMPING HOURS AND CORRESPONDING VOLUMES – DATA OBTAINED FROM MDC.

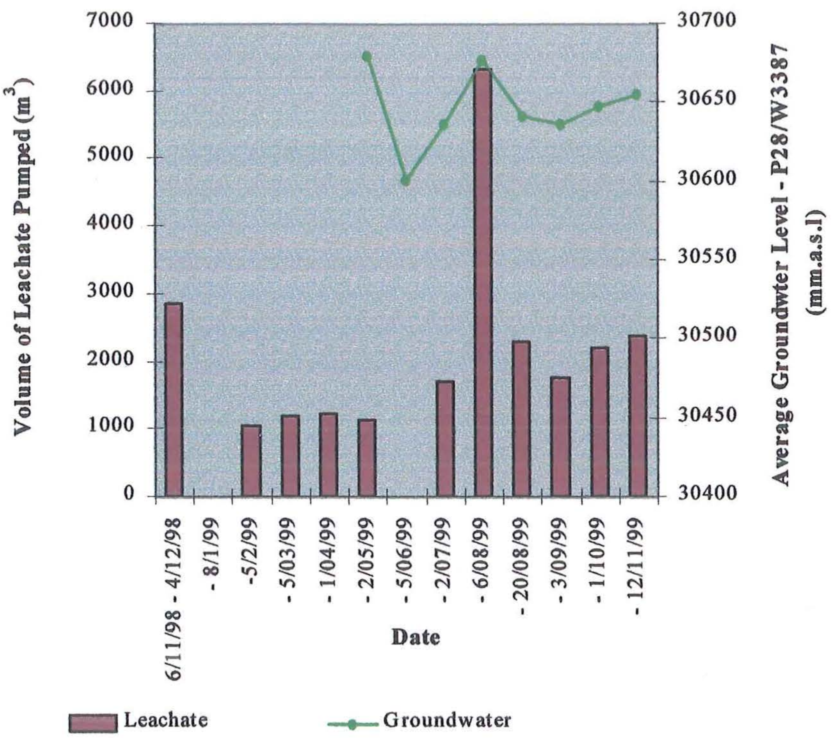
4.5.4 Discussion of Results

Figure 4.22 shows the correlation between pumped leachate volumes, rainfall and groundwater level. From inspection of the results it can be seen that over the period from June 1999 to November 1999 leachate-pumping tend to mirror the trends in groundwater level, suggesting that pumped leachate is predominantly groundwater-sourced. This is further substantiated by the correlation of leachate pumping volumes and total rainfall, which confirms the results of surface water budget calculations. Extremely high leachate pumping volumes observed over the period from 2/0799 to 6/08/99 correspond to a period of high rainfall, which in surface water budget calculations was expected to infiltrate into the landfill. High total rainfall from 1/10/99 to 12/11/99 is not reflected in pumping volumes, as the rainfall was not expected to infiltrate through the landfill cover.

From all available data then it is probable that for the most part, leachate pumped from the perimeter of the landfill is groundwater-generated and given the location of the drainage system with respect to the local groundwater flow, it is likely that the system intercepts some "clean" groundwater and recycles it back through the landfill via the soakage fields. During and/or immediately following the wet winter season rainfall elevates the soil moisture stage capacity above field capacity, and infiltration occurs resulting in the elevation of leachate pumping volumes. Therefore leachate in the southern portion of the landfill can be considered to be largely groundwater derived, except following winter months when leachate-pumping volumes are escalated due to infiltrating precipitation.

The leachate collection system can be assumed to drain approximately 5 ha of the landfill area. Over the 12-month period from November 1998 to November 1999, the total volume of leachate pumped is approximately 24035 m^3 (from Table 4.7). During May 1999, when the main power supply was switched off, it is estimated that approximately 1300 m^3 of leachate would have been pumped based on the surrounding months. Therefore for the entire 12-month period some 25300 m^3 of leachate would have been produced from the 5 ha area at the southern end of the site. Given the surface water budget analysis results of 66 mm infiltration over a similar 12-month period, approximately 3300 m^3 of the leachate produced can be considered to originate from precipitation sources. In the southern portion of the landfill then, the ratio of precipitation to groundwater-sourced leachate is likely to be of the order of 1:7.

Through the middle section of the landfill, the groundwater level remains close to the base of the landfill and can be expected to intercept the base of the landfill due to mounding effects or during periods of elevated groundwater levels. It is therefore expected that leachate produced may be both precipitation and groundwater sourced. The ratio of precipitation to groundwater sourced leachate volume cannot be determined but is expected to be significantly less than in the southern portion of the landfill.



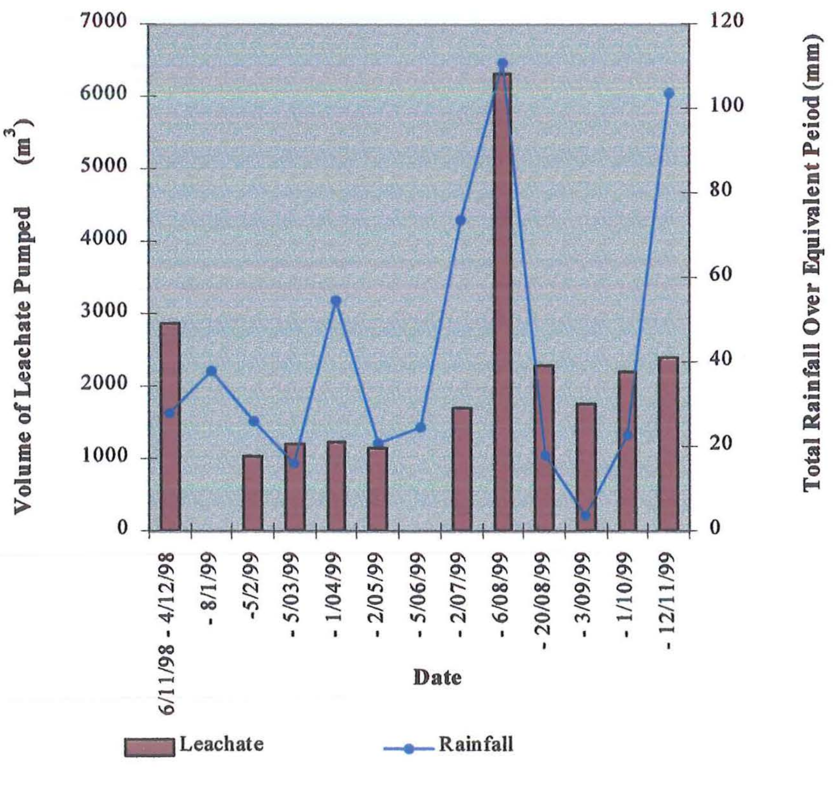


FIGURE 4.22: COMPARISON OF LEACHATE VOLUMES PASSING THROUGH THE PUMPING SYSTEM, TOTAL RAINFALL AND AVERAGE GROUNDWATER LEVEL. NB: TIME PERIODS OVER WHICH PARAMETERS ARE DISPLAYED ARE IRREGULAR, AND ARE DICTATED BY THE TIME OF LEACHATE PUMP READINGS. GROUNDWATER LEVEL IS NOT AFFECTED BY LEACHATE PUMPING.

Over the northern section of the landfill area, groundwater infiltration is expected to be limited as the hydraulic gradient increases and the groundwater level drops significantly below the base of the landfill. Some groundwater mounding may occur but at present the extent of mounding remains undetermined, and all leachate produced is likely to be precipitation-sourced.

4.5.5 Impacts of current utilization

As discussed in Chapters 1 and 2, current utilization of the Taylor Pass Landfill includes the disposal of liquid wastes into open pits within the landfill, whilst repairs and preparations are made to use lined cells at the Regional Blue Gums Landfill. This section gives an indication of the *theoretical maximum input* of liquid from both meteoric sources and liquid waste due to the open nature of the current offal pit site. The resource Consent Application for liquid disposal at the Taylor Pass Landfill site indicates expected volumes of up to 20 m^3 per day. Over a twelve-month period, this corresponds to some 7300 m^3 of hazardous liquid wastes.

Open pits into which the liquid waste is disposed also constitute areas of 100% infiltration of meteoric waters. From Figure 3.15, the area of the landfill occupied by exposed refuse and offal pits is estimated to be of the order of 10,000 m^2. Assuming an annual average rainfall of 600 mm (based on Meadowbank, Blue Gums and Blenheim stations), the volume of meteoric waters infiltrating through the offal pit area alone is 6000 m^3 per annum.

The maximum total amount of liquid entering the landfill as a consequence of continued liquid waste disposal then from both meteoric infiltration and liquid waste disposal is estimated to be approximately 13 300 m^3.

The estimation is meant only as an indication of the maximum possible volumes of liquid input and it is emphasised that the actual liquid input may be less than the calculated volume depending on the available moisture content of the disposed sludges and the effects of evaporation from the open surface of the offal pits.

4.6 Chapter Summary

4.6.1 Main Wairau Aquifer

The main Wairau Aquifer is the uppermost of a series of largely poorly defined aquifers comprised of, and separated by, a series of glacial and interglacial outwash deposits. The aquifer extends over the Wairau Plains from the Waihopai River to Cloudy Bay, with the Wairau River on the northern margin of the Wairau Plains acting as the main source of recharge to the easterly flowing groundwaters. Vertical flow is downward in the unconfined zone to the west of Blenheim, and upward in the zone confined by Dillons Point Formation deposits to the east of Blenheim. Comprising Speargrass and Rapaura Formation gravels, transmissivities in the Wairau Aquifer decrease away from the Wairau River to typically 3000 to 7000 m^2/day in the Blenheim area. The hydraulic gradient over the main Wairau Aquifer of 0.0023 (Davidson *et al.*,1994) results in flow velocities of the order of 0.2 to 2.8 m/day, with preferential paths defined by buried river channels.

4.6.2 Taylor Fan Aquifer

The Taylor Fan Aquifer and other neighbouring aquifer systems recharge the Wairau Plains aquifers from the south. Numerous wells penetrate the distal fan area around New Renwick Road but relatively few production bores are located in the mid to upper portions of Taylor Aquifer and the area south of New Renwick Road has been largely neglected with respect to hydrogeological investigations. Difficulties with well availability and suitability for typical investigation methods have culminated in hydraulic characteristics of the Taylor Fan area being based on extrapolation of data from neighbouring aquifer systems with similar geological histories and deposits. A piezometric survey was carried out over the mid and lower Taylor Fan.

The Rapaura and Speargrass lobes of the Taylor Fan are likely to constitute significantly different hydrogeological environments due to increased fine-grained sediments associated with Speargrass gravels that act to diminish the hydraulic conductivity of the aquifer. Hydraulic conductivity of Rapaura gravels in the mid to upper fan area are expected to be approximately 3×10^{-4} to 8×10^{-4} m/s. The range of hydraulic conductivity of Speargrass formation gravels covers a wider range of 2×10^{-7} to 6×10^{-4} m/s, with the most permeable deposits near the Speargrass/Rapaura boundary.

The piezometric surface in the Taylor Fan area forms a gradient of 1/180 near the southern end of the Taylor Pass Landfill, steepening to approximately 1/50 about 400 m north of the landfill. Piezometric contours indicate that the hydraulic influence of the Taylor Fan Aquifer extend past the geological boundary at New Renwick Road, northeastward to Waters Ave where the dominant flow direction is defined by Wairau groundwaters.

Combining extrapolated hydraulic data and real piezometric data gives possible groundwater flow velocities in the Taylor Fan Aquifer of 3×10^{-4} to 6.9 m/day in the Speargrass Lobe, and 0.5 to 9 m/day in the Rapaura gravels of the modern fan surface. Although these velocities are likely to be possible in the aquifer, true flow velocities are likely to be highly variable due to the heterogeneous nature of Taylor Fan deposits. Buried channels with high permeability and hydraulic conductivity will form preferential flow paths.

4.6.3 Taylor Pass Landfill Hydrology and Hydrogeology

Hydrological and hydrogeological investigations at the Taylor Pass Landfill site were aimed at establishing the extent of infiltration of surface and groundwaters into the refuse body, providing moisture for leachate generation. Surface water infiltration was calculated using the Water Balance

Method (Thornthwaite and Mather, 1955; Fenn *et al.*, 1975), and groundwater infiltration was examined based on local piezometric levels and fluctuations, and landfill perimeter trenching.

Results indicate that over the 1999 year, some 15700 m^3 of precipitation-sourced water percolated through the surface cover of the Taylor Pass Landfill, primarily in July and early August when the field capacity of cover soils was exceeded. Although high rainfall was experienced during November 1999 in the Blenheim area, evapotranspiration processes and soil moisture storage prevented percolation as the moisture capacity of cover soils remained below field capacity. Calculated percolation and anticipated periods of percolation were substantiated by a threefold increase in leachate pumping volumes during July and early August, yet no substantial increase in pumping volumes was noted following high November rainfall. The soil moisture capacity rather than the integrity of the soil or permeability is thought to be the determining factor for percolation rates under the given conditions at the Taylor Pass Landfill. Results from 1999 are thought to demonstrate percolation volumes corresponding to typical annual rainfall.

Evidence suggests that the groundwater table only fluctuates within a 0.5 m range and perennially intercepts the landfill base at the southern end of the landfill site, remaining within 1.5 m of the base through the mid section. The water table drops away from the base of the landfill by up to 5 m in the northern section due to steepening of the hydraulic gradient. In the southern portion of the landfill some 22000 m^3 of leachate produced is thought to be groundwater sourced, with 3300 m^3 sourced from precipitation. The proportion of groundwater to precipitation-sourced leachate is expected to decrease rapidly towards mid and northern sections of the landfill however the installation of piezometers within the landfill would help to clarify the actual extent and rate of groundwater infiltration into the Taylor Pass Landfill.

Chapter 5

Groundwater Quality and Plume Delineation

5.1 Introduction

The concept of *water quality* relates to the physical, chemical and biological features of water and its suitability with respect to a particular use. Water may be considered *polluted* or *contaminated* by a given source when the quality of water down gradient from the potential source is diminished below background quality. Therefore, three major issues must be addressed in contaminated water projects, as follows:

1. The physical, chemical and biological characteristics of both contaminated and uncontaminated waters;
2. The use for which potentially contaminated water is intended or specific expectations people may have;
3. The identification of the true source of contamination, if any, and its distinction from natural background or other contamination sources.

The following chapter investigates the chemical characteristics of groundwater in the main Wairau aquifers and in the Taylor Pass area both upgradient and down gradient of the Taylor Pass Landfill, thus identifying the local trends and contamination sources. As discussed in Chapter 4, groundwater in the Taylor Pass area flows approximately north towards the extensive Wairau aquifer, within which Blenheim Town Supply Bores are located.

5.2 Water Quality Guidelines

Blenheim Town Supply water is intended primarily for human consumption, and as such quality parameters must comply with the New Zealand Drinking Water Standards 1995 (NZDWS) as set out by the New Zealand Ministry of Health (1993). NZDWS sets out maximum concentrations for safe human consumption of chemical, radiological and microbiological contaminants based on current knowledge of health risks. The World Health Organisation (1993) also publishes guidelines for drinking water standards similar to those published by NZMoH. Appendix 5 tabulates data compiled from the World Health Organisation and NZMoH relating to the occurrence and effects of parameters with aesthetic of health based guideline values for which are tested as part of this project and further discussed as part of the following chapter. Not all tested parameters have associated guideline values.

5.3 Wairau Aquifer Chemistry

5.3.1 Main Wairau Aquifer

Rae (1987) describes the groundwater chemistry of the Wairau Plains based on data available to 1987. Close (1994, 1995) presents results of intensive Wairau groundwater surveys conducted in June 1994 and September 1995. Results from the 1994 and 1995 surveys are tabulated in appendix 5.2. Their work forms the basis for the following discussion of Wairau groundwater quality.

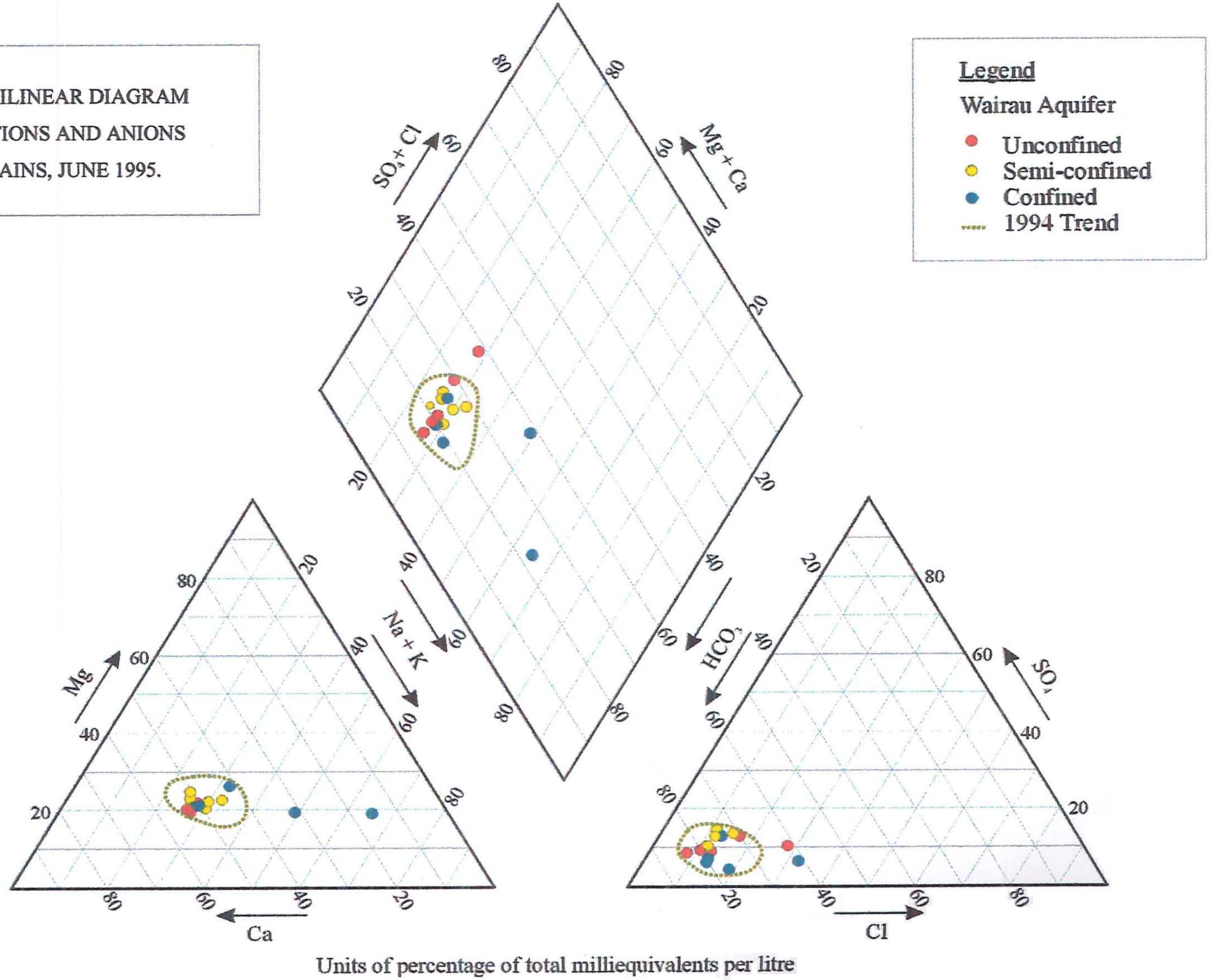
Physical Parameters

Groundwater temperature and conductivity over the Wairau Plains reflect the aquifer recharge pattern described in Chapter 4, and appear to relate primarily to the depth and age of groundwater. Conductivity increases markedly towards the south towards the southern tributary valleys indicating mixing of Wairau waters with more saline waters. The majority of pH data used by Rae (1987) are laboratory-based readings which are commonly altered from field-based readings due to equilibrium reactions taking place between sampling and analysis. No significant pH trends have been determined over the Wairau Plains. The mean pH over the extent of the Wairau Plains is 7.05.

Major Ions

Further analysis by Rae (1987) on chloride/bicarbonate and magnesium/calcium ratios provides additional evidence of a band of "younger" groundwater flowing through the mid-plains region. Figure 5.1 illustrates diagrammatically the trends of major ion data from sampling in May 1994 and June 1995 respectively. Wairau waters are clearly bicarbonate-type waters with the concentration of cations being dominated only slightly by calcium. Such a signature is indicative of young and/or slowly evolving groundwaters (Thorpe, 1992). Samples from confined aquifer sites that do not lie within the general cluster of Wairau results have elevated chloride and/or sodium, and are likely to be affected by either marine sediments associated with confining layers in the coastal reaches of the Wairau Plains which do not extend to the Taylor Fan or mixing with sodium chloride rich Southern Taylor Fan or other tributary waters.

FIGURE 5.1: TRILINEAR DIAGRAM OF MAJOR CATIONS AND ANIONS -WAIRAU PLAINS, JUNE 1995.



Inorganic Contaminants

Rae (1987) identified existing and potential groundwater quality contaminants as manganese and iron. respectively. Manganese concentrations predominantly in excess of drinking water standards (0.05g/m^3) and even over 0.5g/m^3 have little or no spatial pattern, and are likely to be naturally occurring however do remain below levels in the Taylor Fan and levels likely to be expected from a landfill contamination source. Iron concentrations ranging from 0.1-30.0 g/m^3 are reported as being "a rather confusing picture with no immediately obvious regionalisation of the data" (Rae, 1987). More recent sampling has only led to further complexity, as total iron concentrations vary significantly both spatially and over time in the Wairau area (pers. com. P. Davidson, 1999). Testing in 1994 and 1995 included analysis for both iron and manganese; most samples remained below the detection limit.

Nitrate-N levels varied from a maximum level of 2.5 g/m^3 in 1994 to a maximum of 3.6 g/m^3 in 1995, well below the drinking water standard of 11.3 g/m^3.

Other Contaminants

Pesticide contamination was the focus of water quality analyses in 1994 and 1995 surveys. The 1994 survey identified only one well with detectable levels of pesticides in the main Wairau Aquifer. Well P28/w0722 located within the confined Wairau Aquifer at Graham Street contained 0.1 mg/m^3 of 2-phenyl phenol from an undefined source in the 1994 survey, yet none was detected in 1995. No other pesticides were detected in the main Wairau Aquifer in the 1994 or 1995 surveys.

5.3.2 <u>Deep Wairau Aquifer</u>

Investigations into the quality of Deep Wairau Aquifer waters were carried out by Close (1999) and are only briefly discussed here as the confined nature and depth of the aquifers means they are of little relevance to the project at hand. Few penetrating wells and even fewer full chemical analyses means the Deep Wairau groundwater remains poorly characterised at this time. The deep Wairau groundwaters are significantly richer in sodium and chloride than the shallow Wairau counterparts, and may be classified as sodium-bicarbonate-chloride water. Conductivity is consequently also higher than shallow Wairau groundwaters. High levels of alkalinity (140 – 180 g/m^3 as HCO_3) accompany high pH levels (often greater than 8). Nitrate-N, sulphate, calcium and magnesium levels are < 2 g/m^3, < 8 g/m^3, < 20g/m^3 and < 7 g/m^3 respectively. Isotope data suggests water with the order of 17 000 to 27 000 years residence time, and the Wairau and Waihopai Rivers as the main sources of recharge (Taylor, 1999).

5.4 Taylor Fan Groundwater Investigations

5.4.1 Standard Connell Wagner Monitoring Regime

Ground and surface waters in the Taylor Pass Area have been regularly monitored since March 1996 as part of leachate monitoring at the Taylor Pass Landfill. Currently, Connell Wagner Ltd of Blenheim carries out monitoring under contract to the Marlborough District Council.

The monitoring network includes eleven purpose built and pre-existing wells, the leachate sump and the diverted watercourse around the landfill site. Table 5.1 and Figure 5.2 summarise and illustrate details and locations of existing and new monitoring wells and sites. Carried out at six monthly intervals for groundwater bores and 3 monthly intervals for the leachate sump and surface water, Table 5.2 summarises regularly tested parameters.

Hydrocarbons in the form of benzene, toluene, ethyl benzene and xylenes have been tested on a six monthly basis since March 1996. Testing of microbiological contaminants has been deemed unnecessary because the persistence of microbes in migrating groundwater is sufficiently low that it is unlikely that contamination of drinking water sources will be affected.

Well No. (P28/W)	**Location**	**Purpose***	**Collar height** (m.a.s.l.)	**Static water level**** as at 10/06/99 (m)	**Screened interval*** (m)
-	Leachate Sump	R, PS	n/a	n/a	n/a
1313	Eltham Rd	I			above 20.8m?
1477	Redwoodtown School	R		2.77	unknown
2539	Wither Rd (shallow)	R, PS	35.7	4.96	3.0 - 11.3
2540	New Renwick Road	***	27.04	7.37	3.0 - 11.8
2618	Recycling Centre	R	38.49		above 10m?
2619	Marlborough Electric	R	34.28	4.46	above 11.2m?
2661	Arthur Baker Place	R	29.42	7.54	6.0 - 11.8
2662	Riding for the Disabled	R	31.57	6.65	6.0 - 11.8
2663	Wither Rd (deep)	R, PS	35.86	4.88	19.0 - 24.8
3002	Gas Bore	PS	40.06	3.87	6.0 - 11.8
3386	Taylor Farm	PS	52.74	5.76	5.8 - 10.8
3387	Immediately south of landfill	PS	45.39	2.63	14.8 - 19.8
3388	West side of landfill	PS	39.66	1.54	5.2 - 10.2
3389	Page St	PS	28.91	7.51	20.8 - 25.6
3390	Aerodrome Rd	PS	35.18	9.96	13.8 - 18.8
3391	Burleigh Park	PS	27.45	6.74	26.8 - 31.8
CS-IHC	Cleghorn Street	R			unknown

* Regular monitoring network "R", (current) project-specific network "PS" or irregular "I"
** levels quoted below collar *** well no longer tested

TABLE 5.1: TAYLOR PASS LANDFILL MONITORING WELLS.

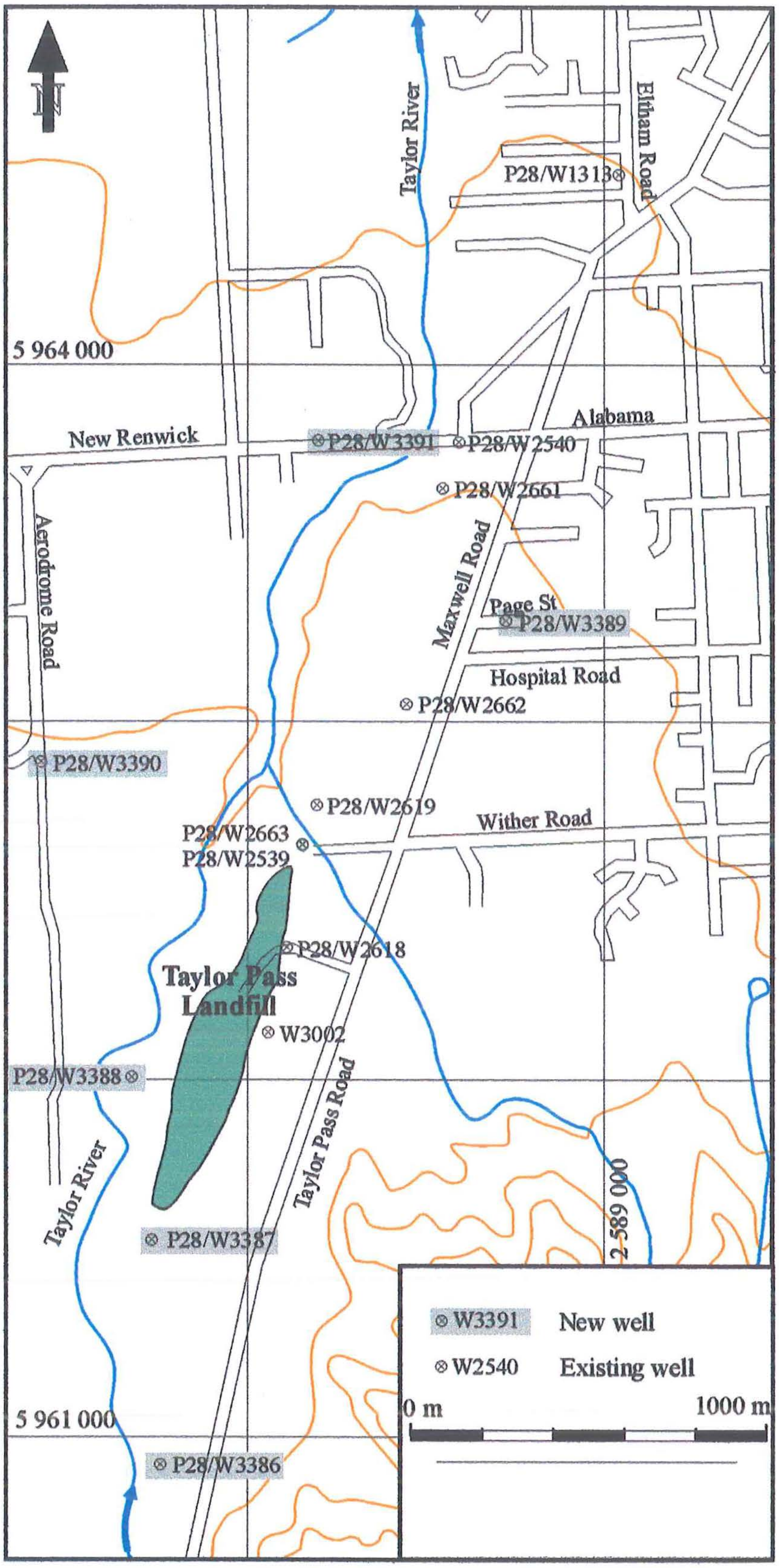


FIGURE 5.2: LOCATION OF TAYLOR PASS LANDFILL MONITORING WELLS

PARAMETER		
	FIELD	DO
		TEMPERATURE
	LABORATORY	CONDUCTIVITY
		pH
		BOD_5
		COD
		SUSPENDED SOLIDS
		AMMONIA-N
		NITRATE-N
		NITRITE-N
		ALUMINIUM
		BORON
		LEAD
		ZINC

TABLE 5.2: TAYLOR PASS LANDFILL MONITORING INDICATORS – STANDARD CONNELL WAGNER MONITORING REGIME.

With respect to the location of the existing monitoring sites, attention must be given to the position of what has been considered prior to the present investigation to be the upstream monitoring bore, P28/W3002 (Figure 5.2). Referred to as the "Gas Bore", the well is located adjacent to the landfill on the eastern side and is likely to be affected by lateral dispersion of leachate from the landfill if not by direct down-gradient flow. As such, the well cannot be considered representative of background Taylor Pass groundwater. The lack of a reliable background well therefore makes the identification of groundwater contaminated by the landfill above the levels of the natural background uncertain from P28/W3002 alone.

Also evident in the standard monitoring network is a distinct lack of wells deeper than 15 m. Well P28/w2663 located at the north end of the landfill on Wither Road is the only deep (>15m) regular monitoring well. Results from P28/w2663 in comparison to shallow monitoring wells (Appendix 5) suggest contamination of groundwater in the vicinity of 19 – 25 m deep at the northern end of the landfill. Results of monitoring at shallow and deep wells are further discussed in following sections. No additional testing of deep groundwaters with respect to contamination from landfill sources is carried out under the standard monitoring regime.

5.4.2 Proposed Tiered Monitoring Regime

In 1998, Connell Wagner proposed a site specific tiered monitoring regime for the regular monitoring of the Taylor Pass Landfill. The three-tiered regime is based on a series of indicator parameters with trigger levels from published water quality standards and guidelines (Table 5.3).

Surface water sites and the leachate sump are monitored for a) first tier parameters on a three monthly basis, b) second tier parameters on an annual basis, and c) third tier parameters on a biennial basis. Groundwater bores are monitored less regularly with a) first tier parameters tested only every six months, and b) second and third tier parameters tested biennially.

If trigger levels are exceeded with respect to two or more parameters at any level, the next tier of parameters is tested. Samples exceeding the trigger levels must also exceed natural background levels of the given parameters to warrant further testing, thus further emphasising the need for a suitable background-monitoring site.

Derivation of the parameters and procedures for implementing the tiered monitoring regime is further discussed with respect to groundwater monitoring bores and the leachate sump only. Surface water contamination, and hence its monitoring, remains beyond the scope of this project.

First tier indicators are based on general parameters and those immediately indicative of contamination from a likely landfill source (e.g. pH, alkalinity, ammonia, chloride, conductivity and TOC), and parameters which been proved problematic in surveys carried out during Resource Consent application works (Royds, 1994) such as iron, manganese and boron. Currently, six monthly monitoring includes second and third tiered parameters of arsenic, chromium, nitrate-N and nitrite-N, yet omits first tier parameters of alkalinity, iron, manganese and TOC.

5.4.3 Project-Specific Monitoring Regime

Derivation and implementation of an intense twelve month, project-specific monitoring regime has been aimed at providing further investigation of deep (>15 m) migration of contaminants, adequate characterisation of background Taylor Pass groundwaters, and the use of first and second tier parameters. Control of the lateral extent of leachate migration and investigation into any seasonal variation of groundwater quality has also been considered. To this end, six new monitoring wells were installed as discussed in section 3.3.2 (Table 5.4 and Figure 5.2).

The available budget enabled monitoring of 10 sites to be conducted bi-monthly over a twelve month period from May 1999. The first, third and fifth monitoring rounds involved testing of first and second tier parameters; the second, fourth and sixth monitoring rounds involved first tier parameter testing only (refer Table 5.3). No additional organic sampling was carried out.

	Trigger Values for Further Monitoring		
	Groundwater bores	**BLM4*, BLM5***	**BLM1***
Field Measurements			
Well depth	Monitor - no trigger	N/A	N/A
Static water level	Monitor - no trigger	N/A	N/A
Dissolved oxygen	80-90% saturation	80-90% saturation	Monitor - no trigger
Temperature	Monitor - no trigger	<2°C increase	Monitor - no trigger
Visual observations	Monitor - no trigger	Monitor - no trigger	Monitor - no trigger
First tier indicators			
Alkalinity	Monitor - no trigger	Monitor - no trigger	Analyse all first tier indicators but do not apply trigger levels.
Ammonia	1.5 mg/l	0.03 mg/l	
Boron*	300 mg/m^3	300 mg/m^3	
Chlouride	250 mg/l	250 mg/l	
Conductivity	150 mS/m	150 mS/m	
Iron (soluble)*	200 mg/m^3	1000 mg/m^3	
Manganese*	500 mg/m^3	500 mg/m^3	
pH	6.5 - 8.5	6.5 - 9.0	
Total organic carbon	Monitor - no trigger	Monitor - no trigger	
Suspended solids	Not monitored	<10% seasonal change	Not monitored
Second tier indicators			
Aluminium	150 mg/m^3	<5 mg/m^3 if pH < 6.5, <100 mg/m^3 if pH>6.5	Analyse second tier indicators if further monitoring is required for any groundwater bores, BLM4 or BLM5. Do not apply triggers
Arsenic	10 mg/m^3	50 mg/m^3	
Bicarbonate	Monitor - no trigger	Monitor - no trigger	
Calcium	Ca + Mg = 200 mg/l	Ca + Mg = 200 mg/l	
Carbonate	Monitor - no trigger	Monitor - no trigger	
Lead	10 mg/m^3	5 mg/m^3	
Magnesium	Ca + Mg = 200 mg/l	Ca + Mg = 200 mg/l	
Nitrogen (total)	Monitor - no trigger	Monitor - no trigger	
Potassium	Monitor - no trigger	Monitor - no trigger	
Phosphorus (total)	Monitor - no trigger	Monitor - no trigger	
Sodium	200 mg/l	200 mg/l	
Sulphate	250 mg/l	250 mg/l	
Turbidity	Not monitored	<10% seasonal change	
Third tier parameters			
Acidity	Monitor - no trigger	Monitor - no trigger	Analyse third tier indicators if further monitoring is required for groundwater bore, BLM4 or BLM5. Do not apply triggers
Cadmium	3 mg/m^3	0.2 mg/m^3	
Chromium	50 mg/m^3	10 mg/m^3	
Cobalt	Monitor - no trigger	Monitor - no trigger	
Copper	2000 mg/m^3	5 mg/m^3	
Dissolved reactive phosphorus	Monitor - no trigger	Monitor - no trigger	
Hydrocarbons (use GC-FID analysis)	Monitor - no trigger	Monitor - no trigger	
Mercury	20 mg/m^3	0.1 mg/m^3	
Nickel	20 mg/m^3	150 mg/m^3	
Nitrate-N	50 mg/l as NO_3	50 mg/l as NO_3	
Nitrite-N	3 mg/l as NO_2	3 mg/l as NO_2	
Zinc	3000 mg/m^3	50 mg/m^3	

* BLM4 and BLM5 are surface water sampling sites. BLM1 is the leachate sump.

TABLE 5.3: PROPOSED TAYLOR PASS LANDFILL MONITORING INDICATORS (FROM CONNELL WAGNER, 1998)

Well number	Location	Depth (m)	Screened interval (m)	Purpose
3386	South of landfill	11	6.0 - 11.0	Background monitoring
3387	Immediately south of landfill	20	15.0 - 20.0	Background monitoring. Fitted with pressure transducer and data recorder.
3388	West of landfill	10	5.0 - 10.0	Detection of westwards migration.
3389	Page Street	26.9	21.9 - 26.9	Detection of deep migration
3390	Aerodrome Road	19.4	14.4 - 19.4	Detection of westwards migration.
3391	Burleigh Park	32	27.0 - 32.0	Detection of deep migration

TABLE 5.4: NEW MONITORING BORES.

Well No.	Location	Sampling Round									
		1 (06/05/99)	**CW** (05/99)	**2** (15/07/99)	**CW** (08/99)	**3** (16/09/99)	**4** (04/11/99)	**5** (17/01/00)	**CW** (02/00)	**6** (29/03/00)	(29/06/00)
-	Leachate Sump (BLM-1)	✓*	✓	✓	✓	✓	✓	✓	✓	✓	✓
1313	Eltham Rd	✓			✓				✓		
1477	Redwoodtown School				✓						
2539	Wither Rd (shallow)			✓	✓	✓	**	**	✓	✓	✓
2540	New Renwick Road										✓
2618	Recycling Centre				✓				✓		
2619	Marlborough Electric				✓				✓		
2661	Arthur Baker Place				✓				✓		
2662	Riding for the Disabled				✓				✓		
2663	Wither Rd (deep)	✓		✓	✓	✓	✓	✓		✓	✓
3002	Gas Bore	✓		✓		✓	✓	✓		✓	✓
3386	Taylor Farm	✓		✓		✓	✓	✓			✓
3387	Immediately south of landfill	✓		✓		✓	✓	✓		✓	✓
3388	West side of landfill	✓		✓		✓	✓	✓		✓	✓
3389	Page St	✓		✓		✓	✓	✓		✓	✓
3390	Aerodrome Rd	✓		✓		✓	✓	✓		✓	✓
3391	Burleigh Park	✓		✓		✓	✓	✓		✓	✓
CS-IHC	Cleghorn Street								✓		

* sample taken as part of Connell Wagner's standard monitoring regime.
** unable to be sampled due to tampering with padlock on well head.

TABLE 5.5: COMBINED MONITORING PROGRAM FROM MAY 1999 TO JUNE 2000.

Table 5.5 summarises sampling undertaken for both the standard and project specific regimes, over the period from May 1999 to June 2000. Results from both the past and current monitoring programmes are tabulated in Appendix 5 and discussed below.

5.4.4 Sampling and Analysis Procedures

Connell Wagner has carried out groundwater sampling and monitoring to date under contract to the Marlborough District Council. In order to eliminate discrepancies caused by different sampling methods and ensure comparable results, Connell Wagner personnel have also carried out water sampling for this project as follows:

1. The static water level in each well is taken prior to pumping.
2. The standing volume of the well is pumped, although recharge rates in the Taylor Pass Area are often slow, causing difficulty in pumping the standing volume without pumping the well dry. Field sheets in Appendix 5 indicate where this problem has restricted pumping volumes. In order to compare the effect of partial pumping on sample quality, all wells were pumped extensively 24 hours before the January sampling round in addition to regular pumping carried out immediately prior to sampling.
3. Samples are collected using a Grundfos® groundwater monitoring submersible pump. Sufficient sample is obtained to carry out first, second and third tier testing if required. Bottles used for the collection of samples are laboratory supplied and all necessary precautions are taken to avoid contamination from other sources and/or cross contamination of samples (i.e. all equipment is washed between samples, and sample bottles are kept sterile).
4. Samples are transported in sealed chilly bins to Cawthron Institute for analysis. Analysis procedures are summarized in Appendix 5.

5.4.5 Assumptions for Water Quality Analysis

In the interpretation of monitoring results, the following assumptions have been made:

1. The water sampling method is reliable and adequate quality control exists with respect to sampling apparatus and cross contamination of samples.
2. Results are indicative of the true nature of water present in the vicinity of the tested well, and there are no extraneous water sources present.
3. Sampling frequency is sufficient to identify changes in water chemistry over time, and any short-term variability (e.g. due to seasonal effects).

5.5 Taylor Pass Groundwater Monitoring Control

5.5.1 Philosophy

In order to identify and assess the effects of leachate from the Taylor Pass Landfill on down-gradient groundwater, it is first necessary to adequately define the quality of background groundwater in the Taylor Pass Area as discussed in Sections 5.1 and 5.3. Maximum background levels must be established as below these levels water down-gradient of the Taylor Pass Landfill cannot be automatically considered to be contaminated by the landfill. Anomalous or elevated parameters detected in background groundwaters cannot be used as indicators of leachate contamination in down-gradient groundwater, thus such parameters must be identified prior to interpretation of down-gradient monitoring results.

Background Taylor Pass groundwater quality analyses are based on 5 suites of samples from P28/w3386 (July 1999 to June 2000, excluding March 2000 when sampling was impossible), 6 suites of samples from P28/w3387 (July 1999 to June 2000), and results of four analyses from regular monitoring of a Regional Blue Gums landfill control bore, BGBH-1 (MDC reference P28/W3134), carried out in 1999. The use of BHBG-1 in conjunction with P28/W3386 and P28/W3387 gives an indication of the spatial variability of background water chemistry down the Taylor Fan surface.

BGBH-1 forms the background control of the monitoring network for the Regional Blue Gums Landfill and is located on the floor of the Taylor Valley approximately 1.8 km south of the Taylor Pass Landfill and 0.7 km west of the Blue Gums landfill (Figure 5.2). BGBH-1 is not affected by contamination from the Blue Gums landfilling site (pers com. A Sweeney – Connell Wagner, 1999). Results from P28/W1313 (Eltham Road) are discussed in Section 5.5.4 to clarify the origin of groundwater which is realistically assumed to be Wairau-derived for comparison with other bores down-gradient of the Taylor Pass Landfill near the boundary between the Taylor and Wairau Aquifers.

5.5.2 Blue Gums Monitoring Well

The average concentration and range of parameters tested at BGBH-1 over the 1999 year are given in Table 5.6. Results indicate a slightly alkaline pH (7.1-7.3) and corresponding moderate alkalinity of 150 g/m^3 as $CaCO_3$. A relatively high conductivity for freshwater of up to 130 mS/m in BGBH-1 reflects high Na and Cl of 200 and 285 g/m^3 respectively. Other major ions tested are SO_4 and K, with concentrations of 1.05 and 1.8 g/m^3 respectively. Only one analysis has been carried out for K hence there is no control over its fluctuation range. Sulphate levels are variable.

Nitrate-N levels of 0.08 g/m^3 are well within the maximum drinking water level of 11.3 g/m^3, and also well within the common range for New Zealand waters of between 0.01 and 5 g/m^3 (Hoare and Rowe, 1992). Nitrite-N remains at undetectable levels. The ranges of ammonia-N levels detected bracket the maximum acceptable value of 1.5 g/m^3, and maximum values detected reach 1.9 g/m^3.

Arsenic, chromium and lead were all tested only once in 1999, with detected levels of 0.035, 0.002 and 0.006 g/m^3 respectively. Copper, nickel and zinc are all below their respective maximum acceptable values, with concentration ranges of 0.005-0.4 g/m^3, <0.002-0.006 g/m^3 and <0.005-

		BGBH-1		P28/W3386		P28/W3387	
Parameter	**units**	**Average**	**Range**	**Average**	**Range**	**Average**	**Range**
DO	ppm			4.43	3.75-5.65	1.97	1.04-2.77
Temp.	°C			15.5	14.7-17.2	14.3	13.5-15.3
Conductivity	mS/m	124	118-130	61.55	61.0-62.7	52.5	49.8-61.2
pH	pH units	7.2	7.1-7.3	7.3	7.0-7.5	7.3	7.0-7.5
Alkalinity	g/m^3 as Ca CO_3	150		195	150-250	120	110-130
Hardness	g/m^3 as Ca CO_3	133	120-150				
Bicarbonate	g/m^3			213	170-240	150	140-160
Carbonate				<1			<1
COD	g/m^3	13	6- 21				
Ammonia-N	g/m^3	1.4	0.5-1.9	0.08	0.047-0.13	0.16	0.13-0.24
Nitrate-N	g/m^3	0.08	<0.01-0.18				
Nitrite-N	g/m^3	<0.001					
Total N	g/m^3			4.2	3.2-5.2	0.27	0.18-0.36
Total P	g/m^3			1.8	1.5-2.1	0.72	0.24-0.97
Chloride	g/m^3	285	260-310	101	86-120	84	81-85
Sulphate	g/m^3	1.05	<0.5-2.1	1.0	0.8-1.3	0.37	<0.005-1.1
Potassium	g/m^3	1.8		4.7	2.6-6.2	1.2	1.0-1.4
Sodium	g/m^3	200		64	62-66	47.5	46-49
Calcium	g/m^3			60	46-78	39	38-41
Magnesium	g/m^3			15	12- 20	8.7	8.3-9.3
Aluminum	g/m^3			0.68	<0.02-2	0.8	<0.02-2.2
Arsenic	g/m^3	0.035		0.009	0.005-0.014	0.05	0.041-0.064
Boron	g/m^3	8.6	7.7-9.5	1.94	1.8-2.1	1.2	1.1-1.2
Chromium	g/m^3	0.002					
Copper	g/m^3	0.018	0.005-0.4				
Iron (soluble)	g/m^3	6.05	2.2-9.9				
Iron (total)	g/m^3			7.2	0.43-22	2.05	0.4-5.2
Manganese	g/m^3	0.65	0.57-0.73	1.14	0.38-2.6	0.49	0.39-0.59
Nickel	g/m^3	0.0	<0.002-0.006				
Lead	g/m^3	0.006		0.06	<0.001-0.13	0.035	0.03-0.039
Zinc	g/m^3	0.021	<0.005-0.028				
TOC	g/m^3			6.6	1.2-16	1.5	1.1-2.0

Maximum value defines background groundwater quality control

TABLE 5.6: TAYLOR PASS BACKGROUND GROUNDWATER QUALITY – SUMMARY OF RESULTS FROM BGBH-1 (1999) AND WELLS P28/W3386 AND P28/W3387 (JULY 1999 - JUNE 2000).

0.028 g/m^3 respectively. Soluble iron values are variable, at 2.2 to 9.9 g/m^3, well in excess of drinking water standards of 0.2 g/m^3. Manganese levels in excess of drinking water standards of 0.5 g/m^3, with concentrations of between 0.57 and 0.73 g/m^3 detected, remain below values recorded in landfill leachate from the Taylor Pass Landfill (Section 5.6.2). Manganese occurs naturally in igneous and metamorphic rock-forming minerals and on the seafloor as manganese nodules (Chang, 1994; Boggs, 1995). The origin of high manganese levels in the Taylor Pass area is unknown, yet spatially variable high concentrations (up to 0.5 g/m^3) are also noted in Wairau groundwaters (Rae, 1987).

Boron is of the order of 30 times the drinking water standard of 0.3 g/m^3 in BGBH-1, with detected levels of up to 9.5 g/m^3. These values are well in excess of boron results in any down-gradient monitoring bores. The origin of naturally occurring boron at such elevated levels negates the parameter as a potential indicator of leachate and leads to questions regarding the boron source.

Wells and Whitton (1977) found boron in New Zealand greywacke ranging from 4.5 to 40 ppm for sandstone components, and 14 to 125 ppm for siltstone and argillite components. Clays removed from soils of greywacke and greywacke-loess parentage also contain levels of boron up to 60 ppm. It is thought that high boron levels in BGBH-1 water may be attributable to a combination of leaching from greywacke basement, dissolution of boron from greywacke derived clays and concentration of dissolved chemical constituents due to high evapotranspiration on the Taylor Fan surface.

5.5.3 Background Monitoring Wells P28/W3387 and P28/W3386

P28/W3387 and P28/W3386 are located 90 and 800 m up-gradient of the Taylor Pass Landfill on the eastern side of the Taylor River. Table 5.6 gives the average concentration and range of parameters tested at wells P28/W3386 and P28/W3387 from July 1999 to June 2000. Results show a neutral to slightly alkaline pH (7.0-7.5) similar to BGBH-1. Dissolved oxygen (DO) reflects the different depths of wells (refer Tables 5.1 and 5.4), with DO levels in the shallow P28/W3386 averaging 4.4 ppm compared to 19.7 ppm in P28/W3387. Alkalinity in P28/W3386 of up to 250 g/m^3 as $CaCO_3$ is due primarily to the bicarbonate ion, with concentrations as high as 240 g/m^3. Both alkalinity and bicarbonate levels drop closer to the landfill in P28/W3387. Likewise, K, Mg and Ca peak in P28/W3386 with recorded maximum levels of 6.2, 20, and 78 g/m^3 indicating an overall increase in ionic concentration in an upgradient direction from P28/W3387 to P28/W3386.

Carbonate concentrations remain below detection levels in wells 3387 and 3386 (excluding the anomalous May 1999 results). Ammonia-N, conductivity, Cl, SO_4, Na and B all fall below levels indicated in BGBH-1, which must thus be considered as background threshold values for these parameters. Aluminium and As levels reach a maximum in P28/W3387, with peak concentrations of 2.2 and 0.064 g/m^3 respectively. Aluminium is highly variable, with levels below detection limits of 0.02 g/m^3 in July 1999 monitoring round. Fe (total) is highly variable in well P28/W3386, with presumably naturally occurring levels ranging from 0.43 to 22 g/m^3 indicating that the iron is not sufficiently stable in background groundwaters to enable the parameter to be used as an indicator for leachate migration. Manganese is elevated above BGBH-1 yet levels of up to 2.6 g/m^3 in P28/W3386, still remain below levels detected in Taylor Pass Landfill leachate which are consistently above 4.3 g/m^3 (Section 5.6.2). Maximum background lead concentrations of 0.13 g/m^3 occur in P28/W3386.

Nutrient indicators of total N and total P are also at a maximum in P28/W3386, with levels of up to 5.2 and 2.1 g/m^3 respectively likely to be attributable to the agricultural nature of the site. Total organic carbon levels of up to 16 g/m^3 in P28/W3386 are highly variable (range 1.2 to 16 g/m^3) and have no obvious source, but their presence at such levels in background waters sets the standard for analysis of groundwaters down-gradient of the Taylor Pass Landfill

5.5.4 Wairau Groundwater

The Blenheim town supply bore P28/W1313 in Eltham Road is the first town-supply down-gradient of the Taylor Pass Landfill, and draws water from the Main Wairau Aquifer. The well has been monitored, generally on a six monthly basis (refer Appendix 5), since August 1996 and once as a part of the monitoring regime for this project (refer Table 5.5). The average and range of results obtained since 1996 are given in Table 5.7. Results indicate slightly acid pH (6.1 average), with corresponding low alkalinity of 53 g/m^3 as $CaCO_3$ in contrast to more alkaline Taylor Pass groundwaters

Ionic concentrations are reduced in the Wairau Aquifer in comparison to Taylor Pass waters, with the cations Ca, Mg, Na and K detected at average levels of 13, 3.8, 10 and 0.96 g/m^3 respectively. Major anions of HCO_3, SO_4, Cl and CO_3 have concentrations of 65, 4.7, 6.5, and <1 g/m^3. Major ion equivalences from BGBH-1, P28/W3386, P28/W3387 and P28/W1313 are displayed graphically in Figure 5.3, which illustrates the drop in ionic concentration from Taylor Pass to Wairau-derived groundwaters. Comparative data for BGBH-1 is incomplete, yet levels of Cl and Na are significantly higher than P28/W3386 and P28/W3387, and comparative stiff plots therefore show a significantly greater ionic concentration and a more marked contrast with Wairau groundwaters.

		P28/W1313	
Parameter	**units**	**Average**	**Range**
DO	ppm	6.1	5.2-7.6
Temp.	0C	13.8	11.0-16.8
Conductivity	mS/m	15	14-18
pH	pH units	6.6	6.5-6.9
Alkalinity	g/m^3 as $CaCO_3$	53	
Bicarbonate	g/m^3	65	
Carbonate	g/m^3	<1	
COD	g/m^3	<5	
Ammonia-N	g/m^3	0.042	<0.005-0.25
Nitrate-N	g/m^3	0.56	0.38-0.73
Nitrite-N	g/m^3	<0.005	
Total N	g/m^3	0.62	
Total P	g/m^3	0.029	
Chloride	g/m^3	6.5	5.7-8.8
Sulphate	g/m^3	4.7	
Potassium	g/m^3	0.96	0.93-0.99
Sodium	g/m^3	10	
Calcium	g/m^3	13	
Magnesium	g/m^3	3.8	
Aluminum	g/m^3	0.06	0.05-0.06
Arsenic	g/m^3	<0.001	
Boron	g/m^3	0.11	0.065-0.2
Chromium	g/m^3	<0.001	
Copper	g/m^3	0.001	
Iron (total)	g/m^3	1.7	
Manganese	g/m^3	<0.01	
Nickel	g/m^3	<0.005	
Lead	g/m^3	0.009	0.001-0.016
TOC	g/m^3	0.6	
Pesticides	mg/m^3	nd*	
Benzene	mg/m^3	<2.5	
Toluene	mg/m^3	<5	
Ethyl benzene	mg/m^3	<2.5	
Xylenes	mg/m^3	<2.5	

TABLE 5.7: P28/W1313 ELTHAM ROAD GROUNDWATER QUALITY – WAIRAU AQUIFER.

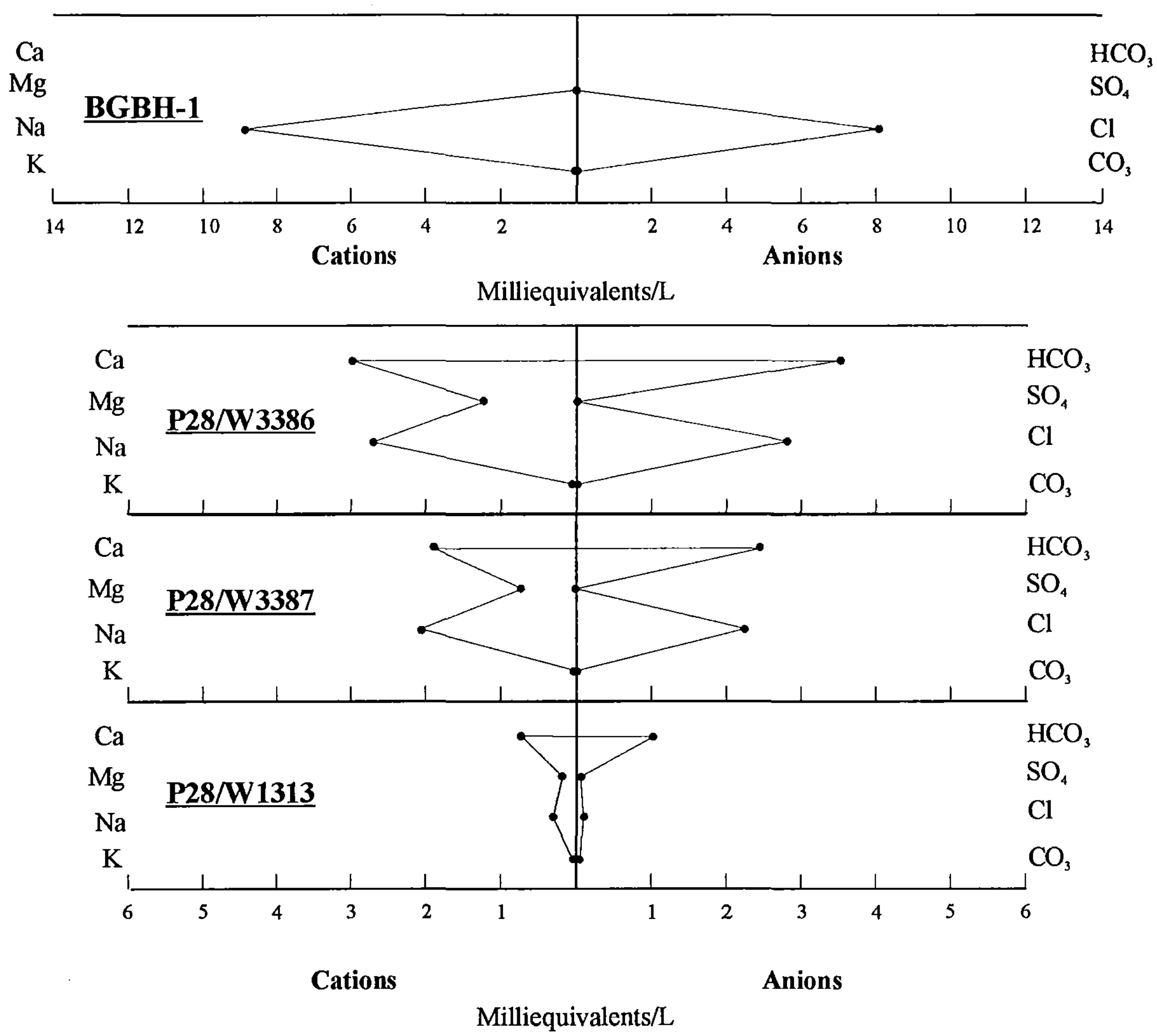


FIGURE 5.3: STIFF PLOTS OF MAJOR IONS IN BACKGROUND TAYLOR PASS AND WAIRAU GROUNDWATERS. NOTE: DIFFERENT SCALE USED FOR BGBH-1. BGBH-1 DATA SET IS INCOMPLETE WITH RESPECT TO CALCIUM AND BICARBONATE.

Dissolved oxygen levels in P28/W1313 of 5.2 to 7.6 ppm and undetectable COD indicate little oxidation activity in the Wairau waters. This is reflected by overall low and often undetected metal and metalloid concentrations. Chromium, Mn and Ni all remain undetected, as does As, however Rae (1987) reported manganese levels of up to 0.5 g/m^3 which must be used as a possible upper limit for Wairau groundwaters for further interpretation. Aluminium has only been tested twice, yet the values of 0.05 and 0.06 g/m^3 suggest that aluminium is more stable at Eltham Road than on the Taylor Fan. Boron levels are lower than background Taylor Fan levels, but appear still to be variable with levels of 0.065 to 0.2 g/m^3 reported. Total iron has only been tested

once and given the variability reported on the Taylor Fan above and also across the Wairau Plains by Rae (1987), the single reported level of 1.7 g/m^3 at Eltham Road cannot be taken as a true representation of iron levels. Lead and Cu both remain below Taylor Fan levels, with concentrations of 0.001 and 0.001-0.016 g/m^3 respectively. Conductivity of Wairau groundwaters is stable at low values between 14 and 18 mS/m.

Pesticides and common hydrocarbons (benzene, toluene, ethyl benzene and xylenes) tested as part of the Connell Wagner monitoring regime remain undetected, however pesticide levels have not been tested since February 1997. The lack of hydrocarbons is reflected in a low TOC result of 0.6 g/m^3. Nitrogen in the form of NH_4, NO_2 and NO_3 all remain low, with total N levels of 0.62 g/m^3. Total P is of the order of 0.3 g/m^3.

5.5.5 Key Conclusions

Main conclusions drawn from analysis of results for Taylor Pass Landfill control bores BGBH-1, P28/W3386, P28/W3387 and P28/W1313 are as follows:

- P28/W1313 can be considered to represent Wairau-derived groundwaters given that concentrations of tested parameters are similar to those reported by Rae (1987) and as discussed in Section 5.3.
- All tested parameters decrease in concentration from the Taylor Fan to Wairau-derived groundwaters. This is a likely effect of high evapotranspiration and minimal recharge acting to concentrate dissolved chemical constituents in the Taylor Fan area in comparison to the main Wairau Aquifer, which is recharged from the Wairau River.
- On the basis of the variability of naturally occurring levels in control bores, boron and iron are likely to be unreliable indicators for the identification of leachate contamination in down-gradient groundwaters. Further discussion of B and Fe concentrations is therefore deemed unnecessary with respect to leachate contamination and is omitted from the remainder of this project. Further investigation of the origin of these extraordinary parameters is, however, recommended for future work.
- Parameters which are potential indicators for control of leachate investigations are listed in Table 5.8, along with a) maximum Taylor Pass background levels above which down-gradient bores may be considered contaminated, and b) maximum concentrations of constituents in Wairau groundwater, below which groundwater may be considered to be either entirely Wairau derived or substantially diluted by mixing with Wairau groundwater. Dissolved Oxygen and pH limits are given as ranges rather than maximums due to the very nature of the parameters.

Parameter	units	Maximum Taylor Pass Background Levels	Maximum Wairau Aquifer Levels
DO	ppm	1.04-5.65	5.2-7.6
Conductivity	mS/m	130	18
pH	pH units	7.0-7.5	6.5-6.9
Alkalinity	g/m^3 as Ca CO_3	250	53
Bicarbonate	g/m^3	240	65
Carbonate	g/m^3	<1	<1
COD	g/m^3	21	<5
Ammonia-N	g/m^3	1.9	0.25
Nitrate-N	g/m^3	0.18	0.73
Nitrite-N	g/m^3	<0.001	<0.005
Total N	g/m^3	5.2	0.62
Total P	g/m^3	2.1	0.029
Chloride	g/m^3	310	8.8
Sulphate	g/m^3	2.1	4.7
Potassium	g/m^3	6.2	0.99
Sodium	g/m^3	200	10
Calcium	g/m^3	78	13
Magnesium	g/m^3	20	3.8
Aluminum	g/m^3	2.2	0.06
Arsenic	g/m^3	0.064	<0.001
Chromium	g/m^3	0.002	<0.001
Copper	g/m^3	0.4	0.001
Nickel	g/m^3	0.006	<0.005
Lead	g/m^3	0.13	0.016
Zinc	g/m^3	0.028	
Manganese	g/m^3	2.6	0.5*
TOC	g/m^3	16	0.6

TABLE 5.8: CONTROL DATA FOR THE ASSESSMENT OF LEACHATE CONTAMINATION OF GROUNDWATERS DOWN-GRADIENT OF THE TAYLOR PASS LANDFILL. *BASED ON DATA FROM RAE (1987)

5.6 Leachate Monitoring Results

5.6.1 Interpretation Approach

Given the limits of background water chemistry levels identified in the previous section it is now necessary to firstly investigates contaminant levels and trends in leachate being actively produced at the Taylor Pass Landfill, and sampled from the leachate sump BLM-1, thus providing a basis for the identification of leachate in groundwater down-gradient of the landfill site. Wells P28/W2539 and P28/W2663 are located immediately down-gradient of the Taylor Pass Landfill and any contamination present in these two wells is assumed to be sourced from the northern end of the landfill and representative of leachate being generated within the older section of the landfill (refer Chapter 1).

Of parameters tested in BLM-1, P28/W2539 or P28/2663, only those that *exceed* background groundwater levels are necessarily considered to be indicators of leachate contamination. Thus leachate indicators are effectively defined by a process of elimination of those parameters that may be attributable to naturally occurring processes or some undefined source of contamination upgradient of the Taylor Pass Landfill. New sources of contamination down-gradient of the Taylor Pass Landfill, such as the former Brayshaw Park Landfill, are not so easily determined unless contaminating parameters are either absent or negligible in Taylor Pass Landfill leachate. The distinction between contamination sources is further discussed in Section 5.6.4, in relation to groundwater quality down-gradient of the Taylor Pass Landfill.

Parameters in excess of background levels due to Taylor Pass Landfill leachate sources are identified in Sections 5.6.2 and 5.6.3 with respect to BLM-1 and immediate down-gradient bores (P28/W2663 and P28/W2539) respectively, and are then compared with groundwater monitoring data from down-gradient monitoring wells in an attempt to identify wells that have been contaminated by the landfill (Section 5.6.4) and to aid in establishing the extent of the leachate plume (Section 5.7).

5.6.2 Taylor Pass Landfill Leachate – BLM-1

Origin of Leachate

Given that the drainage system that feeds the leachate sump is located adjacent to that portion of the landfill which is known to intercept the groundwater table, in-between the landfill and the diversion drain (refer Section 4.5.3), it is likely that a) the leachate produced within the landfill in this area and sampled at BLM-1 is predominantly groundwater sourced, and b) that mixing with groundwater seeping from the diversion drain considerably dilutes sampled leachate and may mask true levels of contaminants in raw leachate still being produced at the site.

An additional sample from BLM-1 was taken on 1 December 1999 following high rainfall in November 1999 (refer Appendix 4) in an attempt to establish the effect of precipitation-derived leachate on the composition of leachate sampled at BLM-1. Subsequent surface water balance analyses discussed in Chapter 4 indicate that precipitation did not actually percolate through the cover material at this time. Consequently results of the additional sampling of BLM-1 only show diluted concentrations of contaminants resulting from increased groundwater seepage due to elevated groundwater levels. Results of the additional December monitoring of BLM-1 are included in Appendix 5, but are omitted from analysis of leachate characteristics. BLM-1 was sampled as part of the Connell Wagner monitoring regime on 20 August, immediately following the period of rainfall percolation into the landfill. This monitoring round is specifically discussed within the following section on leachate composition.

With respect to mixing of the leachate with groundwater prior to sampling at BLM-1, once raw leachate enters the groundwater and migrates away from the site it is effectively diluted by the groundwater with which it mixes, hence although the solution sampled from the leachate sump may not represent true undiluted leachate produced at the site, it is certainly representative of the likely nature of groundwater immediately surrounding the site which is landfill-derived and modified.

Leachate Sampling Frequencies

Monitoring of leachate at the leachate sump, BLM-1, has been carried out by Connell Wagner on a three-monthly basis since March 1996, and on a two-monthly basis specifically for this investigation from July 1999 to February 2000. The following discussion on composition refers to data from both sources obtained from May 1999 to June 2000. Data from 1996 onwards is used to examine long-term compositional trends.

Leachate Composition

The composition of leachate sampled at BLM-1 over the period from May 1999 to June 2000 is summarised in Table 5.9. Elevated conductivity of leachate (215-259 mS/m) reflects the mobile nature of salts present and formed within the refuse body, which are soluble at pH levels below neutral. Measured pH of between 6.8 and 7.3 in BLM-1 is likely to be affected by mixing with groundwater. Actual pH levels within the landfill are likely to be much more acidic due to production of acids during waste stabilisation (refer Chapter 2). On its own pH is of little use in defining leachate given both the numerous variables affecting the parameter and the slightly acid nature of fresh Wairau-derived groundwaters.

The range of dissolved oxygen reflecting redox activity in BLM-1 falls within the limits defined by background levels in Table 5.8, however it is the lower limit of DO in background waters that defines the boundary between uncontaminated and potentially contaminated groundwaters. Where DO indicates the amount of oxygen used in oxidation reactions, COD indicates the *demand* for oxygen for oxidation reactions. COD in leachate at BLM-1 is 133 g/m^3 compared with background demand of 21 g/m^3.

Both sodium and chloride concentrations in BLM-1 are elevated above background levels in wells P28/3386 and P28/W3387 yet remain below high background levels of 200 and 148 g/m^3 detected in BGBH-1. Carbonate levels remain undetected in both background waters and leachate. Concentrations of sulphate, potassium, calcium and magnesium, however, are all elevated above background levels, with concentrations of 23, 103, 135 and 48 g/m^3 respectively. The concentration of sulphate at a maximum of 30 g/m^3 remains well below the maximum acceptable value for drinking water of 250 g/m^3, and is thus not considered to pose a significant risk to down-gradient consumers.

		BLM-1	
Parameter	**Units**	**Average**	**Range**
DO	ppm	3.77	2.77-4.35
Conductivity	mS/m	226	215-259
pH	pH units	7	6.8-7.3
Alkalinity	g/m^3 as Ca CO_3	9.8	860-990
Bicarbonate	g/m^3	1150	1100-1200
Carbonate	g/m^3	<1	
COD	g/m^3	133	120-150
Ammonia-N	g/m^3	66	46-86
Nitrate-N	g/m^3	6.1	2.1-10
Nitrite-N	g/m^3	0.41	0.15-0.66
Total N	g/m^3	84	80-88
Total P	g/m^3	0.48	0.37-0.55
Chloride	g/m^3	148	120-190
Sulphate	g/m^3	23	15-30
Potassium	g/m^3	103	78-150
Sodium	g/m^3	113	100-130
Calcium	g/m^3	135	130-140
Magnesium	g/m^3	48	47-49
Aluminum	g/m^3	<0.02	
Arsenic	g/m^3	0.017	0.016-0.018
Chromium	g/m^3		
Copper	g/m^3		
Nickel	g/m^3		
Lead	g/m^3	0.003	0.001-0.007
Zinc	g/m^3	0.016	
Manganese	g/m^3	4.7	4.3-5
TOC	g/m^3	43	16-66

TABLE 5.9: COMPOSITION OF LANDFILL LEACHATE TESTED AT BLM-1

Bicarbonate is often associated with landfills due to the dissolved inorganic carbon formed during the oxidation of organic matter (Deutsch, 1997), and bicarbonate levels are reported up to 1200 g/m^3 in the Taylor Pass Landfill leachate at BLM-1.

Manganese concentrations of between 4.3 and 5 g/m^3 are lower than a one-off measured value in 1996 of 8.4 g/m^3. Deutsch (1997) identifies manganese as a dominant species in landfill leachate and in a model of leachate migration based on redox potential and retardation of contaminants (Figure 5.4), manganese is a principal component at the leading edge of a theoretical plume. Manganese above background Taylor Pass groundwater levels down-gradient of the Taylor Pass Landfill is expected to be landfill sourced and the extent of migration is likely to be defined by the extent of elevated manganese assuming that the theoretical model presented by Deutsch (1997) is applicable to leachate emanating from the Taylor Pass Landfill.

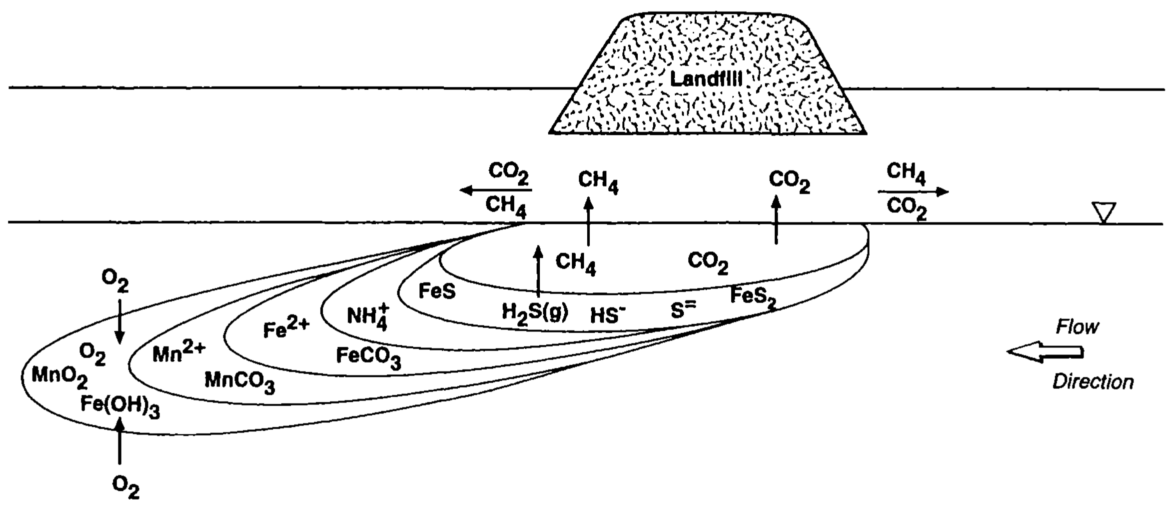


FIGURE 5.4: MODEL OF LEACHATE ZONES DOWN-GRADIENT OF A LANDFILL (FROM DEUTSCH, 1997)

Also forming a dominant zone in the model proposed by Deutsch (1997) is ammonia. Retardation of ammonia due to its attraction to cation exchange sites delays the migration of NH_4 in comparison to the leading edge dominated by manganese. Ammonia-N concentrations are elevated in BLM-1 to a maximum of 86 g/m^3, well above maximum background levels of 1.9 g/m^3. Given that the ammonia-N in leachate is 40-50 times the concentration of background waters, the ammonia-N zone within the leachate plume should be easily recognisable in down-gradient waters and together with manganese should give a reliable indication of the direction and rate of leachate migration.

Chromium, copper and nickel concentrations in BLM-1 have not been tested since March 1996 when concentrations were recorded at 0.73, 0.20 and 0.31 g/m^3 respectively. Chromium and nickel levels tested in 1996 are both above the threshold limit defined by background levels however insufficient testing of all three parameters has been undertaken since March 1996 to accurately clarify its relevance to this study. Chromium, copper and nickel are therefore disregarded with respect to leachate identification.

Total organic carbon is elevated to up to 66 g/m^3 in BLM-1, indicating the likelihood of some form of organic plume possibly comprising either petroleum hydrocarbons forming a LNAPL (light non-aqueous phase liquid) plume or chlorinated solvents forming a DNAPL (dense non-aqueous phase liquid) plume.

As previously discussed, those parameters tested in BLM-1 that remain below the maximum background levels in Table 5.9 may be naturally occurring rather than leachate-derived. However BLM-1 samples leachate only from the southern end of the landfill site and parameters remaining below background Taylor Pass Levels in BLM-1 may be generated as leachate in the southern end of the site and hence evident in immediate down-gradient bores. Rather than immediately eliminate parameters as leachate indicators following assessment of leachate at BLM-1 it is more appropriate to identify a) those parameters which are *probable* indicators of leachate contamination (i.e.

parameters present in BLM-1 at concentrations higher than background), and b) those parameters which are *possible* indicators of leachate contamination (i.e. those parameters which are commonly associated with landfill leachate but are absent or significantly masked in BLM-1). Probable and possible indicators of leachate contamination following the assessment of BLM-1 are listed in Table 5.10.

Probable Leachate Indicators		
Ammonia-N Manganese Dissolved Oxygen Conductivity Alkalinity	Bicarbonate COD Nitrate-N Nitrite-N Total N	Sulphate Potassium Calcium Magnesium TOC
Possible Leachate Indicators		
Carbonate Total Phosphorus Lead	Chloride Sodium	Aluminium Arsenic

TABLE 5.10: PROBABLE AND POSSIBLE INDICATORS OF LEACHATE CONTAMINATION FOLLOWING ANALYSIS OF BLM-1 MONITORING RESULTS.

Long-term Trends in Leachate Composition

Of the parameters identified above as being indicative of leachate contamination, conductivity, ammonia-N, nitrate-N and nitrite-N are the only parameters that have been tested regularly since 1996. Figure 5.5 shows the long-term variation in these parameters in addition to long-term trends of total N calculated as the sum of nitrogen associated with ammonia, nitrate and nitrite. Trends

indicate a decline in conductivity, COD and total N, with total N being dominated by ammonia-N. Nitrate and nitrite nitrogen are unstable with time and show no long-term trend.

Lu *et al.* (1985, in Qasim and Chiang, 1994) devised rate equations and rate constants to predict the decrease in concentration of a number of leachate species with time based on upper concentration boundaries of data from actual landfills, field test cells and laboratory columns. No significant trend was identified by Lu *et. al.,* for the decline in concentration nitrate-N or nitrate-N. Similarly, no trend is evident for these parameters in BLM-1 (refer Figure 5.5). Rate equations for COD, and ammonia N in g/m^3 were however determined as:

- $COD = 89{,}500 \times 10^{-kt}$ where k = 0.0454 (rate constant for COD)

  t = time since closure (years)

- $NH_4\text{-}N = 12{,}000 \times e^{-kt}$ k = 0.10 (rate constant for ammonia-N)

(from Lu *et al.*, 1985 in Qasim and Chiang, 1994).

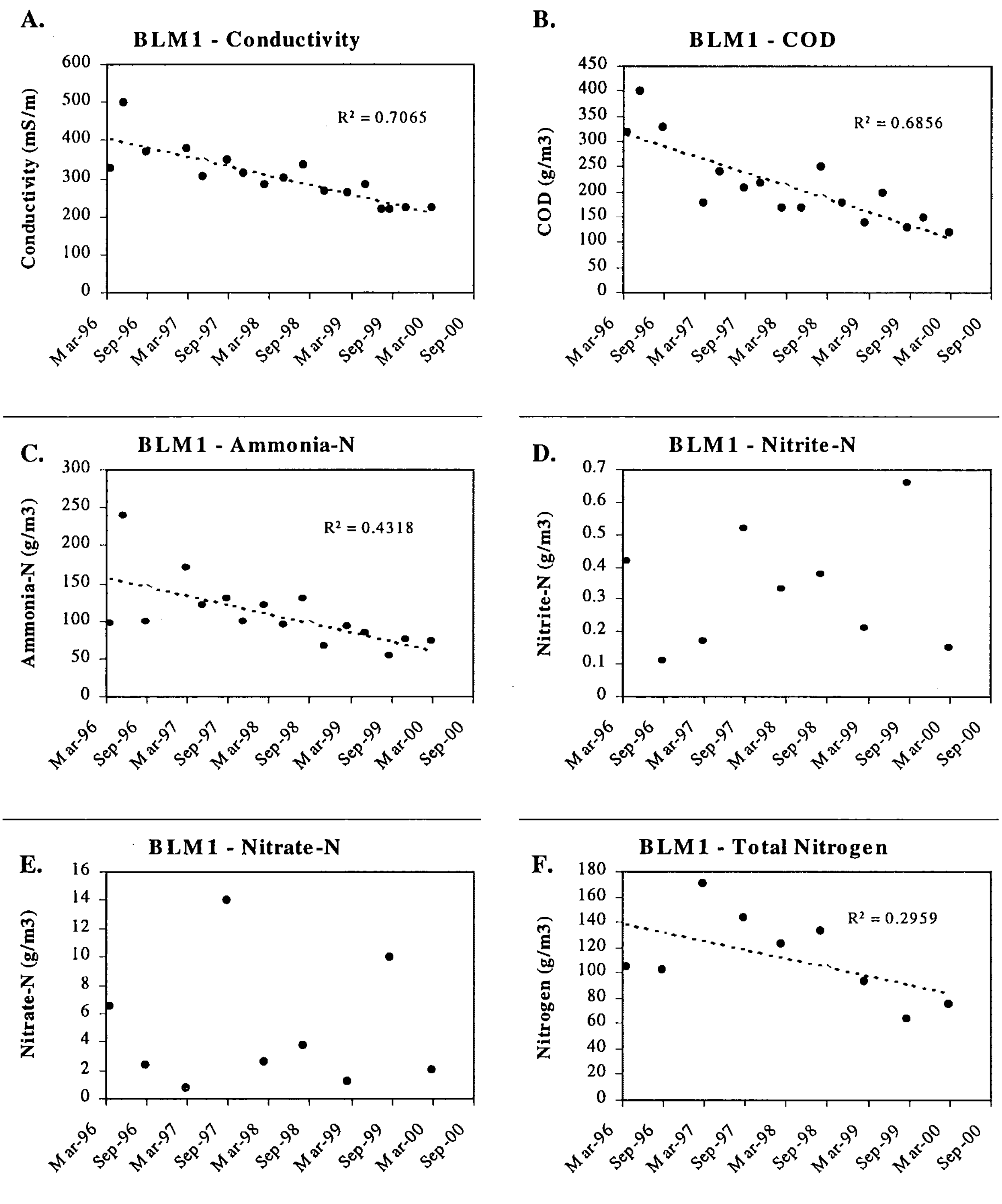


FIGURE 5.5: LONG TERM VARIATION IN LEACHATE CHARACTERISTICS AT BLM-1 A) CONDUCTIVITY (MS/M), B) CHEMICAL OXYGEN DEMAND (G/M^3), C) AMMONIA-N (G/M^3), D) NITRATE-N (G/M^3), E) NITRITE-N (G/M^3) AND TOTAL NITROGEN (G/M^3).

Using a period of 4 years since closure of the landfill, predicted levels of COD and ammonia-N in raw leachate are 59,000 and 8,000 g/m^3 respectively. The predicted levels are well in excess of levels detected at the Taylor Pass Landfill however they do indicate that the concentrations of contaminants leached from a landfill decrease following a negative exponential curve (Figure 5.6), rather than by the adherence to the linear trends indicated in Figure 5.5.

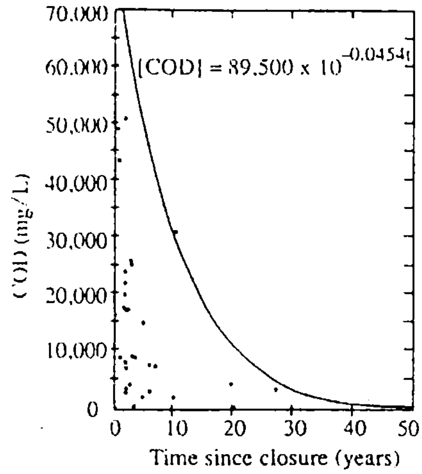


FIGURE 5.6: RATE CURVE FOR COD IN LANDFILL LEACHATE
(AFTER LU *ET AL.* 1985, IN QASIM AND CHIANG, 1994).

Declining levels of COD and conductivity indicate a decrease in total ionic concentrations with time as infiltrating groundwaters progressively leach contaminants from the base of the landfill. Fluctuations in levels in relation to the overall trend may be attributable to changing groundwater levels within the landfill and percolation of precipitation. High groundwater levels are likely to result in increased concentrations of contaminants leached from the refuse body as waste that normally remains above the water table becomes saturated with rising groundwater. Conversely however, the actual measured concentration of leachate at BLM-1 will be diluted by elevated fresh water volumes passing through the drainage system as discussed earlier in this section. By the same reasoning, percolation of precipitation will also increase the ionic strength of leachate produced for a time.

5.6.3 Immediate Down-gradient Wells

Sampling Sites

P28/W2663 and P28/2539 are located off the end of Wither Road, and sample water from 19-24.8 m and 3-11.3 m deep approximately 40 m down-gradient of the Taylor Pass Landfill within Rapaura gravels of the Taylor Fan. Due to their proximity to the landfill in an immediate down-gradient direction, it can be safely assumed that any contamination of groundwater above background levels in both wells originates from leachate produced within the Taylor Pass Landfill and may in reality be more representative than BLM-1 of leachate produced from the older northern section of the landfill. It is necessary then to reinvestigate the concentration of parameters identified in Table 10 as being *possible* indicators of leachate contamination in addition to assessing contamination by *probable* leachate indicators also listed in Table 10.

Water chemistry test results for both P28/W2663 and P28/W2539 over the period from May 1999 to June 2000 are summarised in Table 5.11. Of particular interest with respect to composition of groundwaters in the two adjacent wells is the difference in concentration of contaminants between deep and shallow waters, and the variability of concentrations with time in both wells. These issues are further discussed below.

Composition

With reference to Table 5.11 the following parameters remained below maximum Taylor Pass background groundwater quality levels in both P28/W2663 and P28/W2539 over the period from May 1999 to June 2000:

- Carbonate (<1 g/m^3)
- Total phosphorus (maximum 1.7 g/m^3)
- Sulphate (maximum 1.2 g/m^3)
- Lead (maximum 0.1 g/m^3)

		P28/W2663		P28/W2539	
Parameter	**Units**	**Average**	**Range**	**Average**	**Range**
DO	g/m^3	2.1	0.58-4.68	5.37	2.95-7.23
Conductivity	mS/m	190	126.4-208	73.90	12.4-16.6
pH	pH units	7.2	6.7-7.4	6.6	6.1-7.3
Alkalinity	g/m^3 as Ca CO_3	258	170-660	102.00	48-190
Bicarbonate	g/m^3	340	210-810	356	58-780
Carbonate	g/m^3	<1		<1	
COD	g/m^3				<5-11
Ammonia-N	g/m^3	1.5	0.34-7.4	2.80	0.05-15
Nitrate-N	g/m^3	0.7			<0.005-1.5
Nitrite-N	g/m^3	<0.001			<0.005-0.05
Total N	g/m^3	0.73	0.64-0.78	1.90	
Total P	g/m^3	1.37	1.2-1.7	0.43	0.24-0.62
Chloride	g/m^3	488	11.-590	188	4.3-500
Sulphate	g/m^3		<0.005-1.2	4.2	1.8-6.6
Potassium	g/m^3	2.3	2.2-2.4	8	2- 14
Sodium	g/m^3	245	230-260	36	7.7-65
Calcium	g/m^3	118	110-120	83	15-150
Magnesium	g/m^3	28	25-29	21	4.4-37
Aluminum	g/m^3	1.07	<0.02-3.3		<0.02-0.07
Arsenic	g/m^3	0.10	0.08-0.12		<0.001-0.007
Chromium	g/m^3	0.003		0.0025	0.002-0.003
Copper	g/m^3				
Nickel	g/m^3				
Lead	g/m^3	0.045	0.008-0.1	0.006	
Zinc	g/m^3				
Manganese	g/m^3				
TOC	g/m^3	6	1.2-29	9.1	1.7-30

TABLE 5.11: COMPOSITION OF GROUNDWATER IMMEDIATELY DOWN-GRADIENT OF LANDFILL (FROM MAY 1999 TO JUNE 2000)

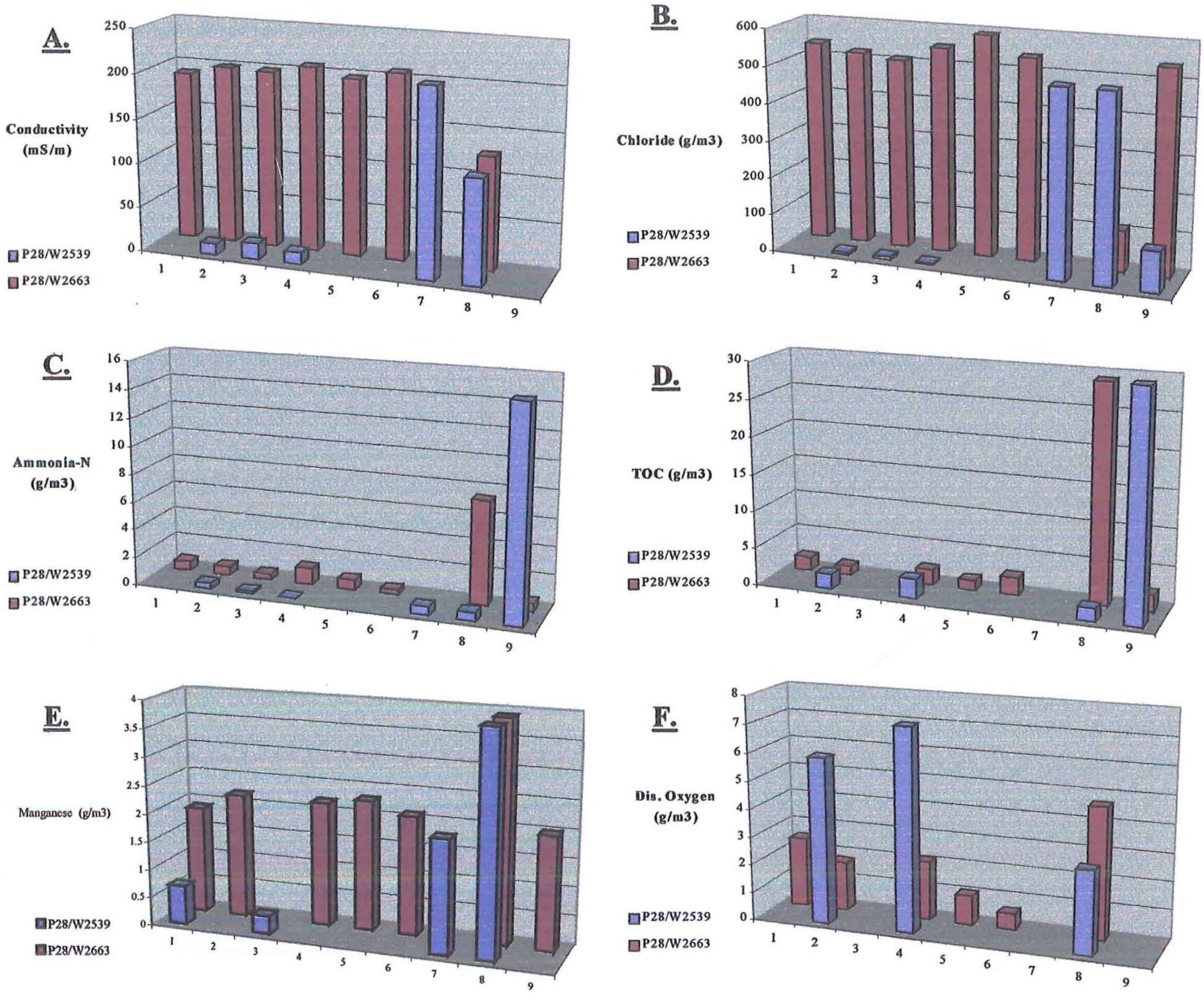
Carbonate, total phosphorus and lead were also noted as not exceeding maximum Taylor Pass background levels when tested in BLM-1, and hence can be eliminated from further investigation. Sulphate was noted as being elevated in leachate and must remain a potential leachate indicator, however its absence above background levels in immediate down-gradient bores through 5 monitoring rounds for P28/W2539 and 7 monitoring rounds for P28/W2663 suggests that it may be a product of younger refuse only, or that it is rapidly attenuated by dilution and anion exchange.

Overall, water chemistry results from wells P28/W2663 and P28/W2539 are highly variable and complex. None of the tested parameters remain above background levels through all monitoring rounds carried out between May 1999 and June 2000. Figure 5.7 shows comparative data for selected parameters over nine monitoring rounds from May 99 to June 2000, although it must be noted that wells have not been tested for all parameters in all monitoring rounds.

Conductivity is closely related to chloride levels and both follow the same general pattern with elevated levels in P28/W2663 (190-208 mS/m and 510-590 g/m^3) and low levels in P28/W2539 (12-18 mS/m and 4-8 g/m^3) through the monitoring rounds to January 2000. Both conductivity and chloride levels diminish in P28/W2663 in March 2000 to 126 mS/m and 110 g/m^3 following an increase in chloride and consequently conductivity to 500 g/m^3 and 209 mS/m in P28/W2539 in February 2000. The presence of elevated chloride and sodium in immediate down gradient P28/W2663 reveals that both parameters are indicators of leachate contamination however they are largely concentrated in deeper waters of P28/W2663, showing up in P28/W2539 in February and March 2000, when they correspondingly decrease in P28/W2663. This trend suggests a complex hydraulic connection between the two wells, which is further discussed later in this section.

Chloride and sodium levels are recorded to approximately 4 times the natural background and are thus considered to represent groundwater contamination from a leachate source. Bicarbonate, ammonia, and TOC all show similar trends to one another with extremely high concentrations in P28/W2663 in March 2000 followed by similar concentrations recorded in P28/W2539 in June 2000. High dissolved oxygen in P28/W2539 in July and September reflects the shallow nature of the monitoring well with oxygen able to be readily transferred at the water table.

Manganese, which could form the leading edge of a plume as indicated in the model presented by Deutsch (1997) remains below background levels except during the March 2000 monitoring round when levels in both wells are in excess of 3.5 g/m^3 indicating, as would be expected, that the maximum extent of the plume has migrated past the P28/W2663 and P28/W2539. The elevated level of manganese in March 1999 corresponds with a similar spike in concentration bicarbonate, ammonia and TOC in P28/W2663. Dissolved oxygen also shows a small spike in March 2000. The combination of elevated parameters may indicate a pulse of leachate passing through P28/W2663 in particular. Elevated DO may suggest the pulse comprises relatively young water in which redox



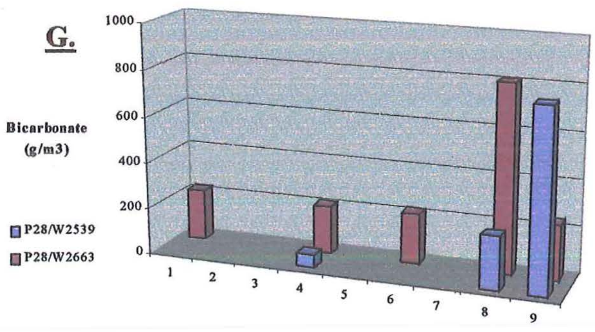


FIGURE 5.7: COMPARATIVE BAR GRAPHS OF WATER CHEMISTRY IN P28/W2663 AND P28/W2539. A) CONDUCTIVITY, B) CHLORIDE, C) AMMONIA-N, D) TOC, E) BICARBONATE, AND G) DISSOLVED OXYGEN. SAMPLING ROUND NUMBERS: 1 = MAY 99, 2 = JUL 99, 3 = AUG 99, 4 = SEP 99, 5 = NOV 99, 6 = JAN 00, 7 = FEB 00, 8 = MAR 00, 9 = JUN 00.

reactions have not yet acted to reduce the oxygen content. The origin of such a water source is currently undetermined and the interpretation then must be remains cautionary.

Stiff plots of major ions (Figure 5.8) indicate that the nature of groundwater in P28/W2539 over the period September 1999 and January 2000 monitoring rounds is analogous to Wairau-derived groundwater, yet in June 2000 the signature completely changes to one resembling the leachate in BLM-1. The implications of this are discussed in the following subsection. In the analysis of long-term trends, P28/W2663 shows generally higher concentrations of contaminants than P28/W2539 (Figure 5.9) with the exception of ammonia-N, suggesting that the majority of the plume body, and

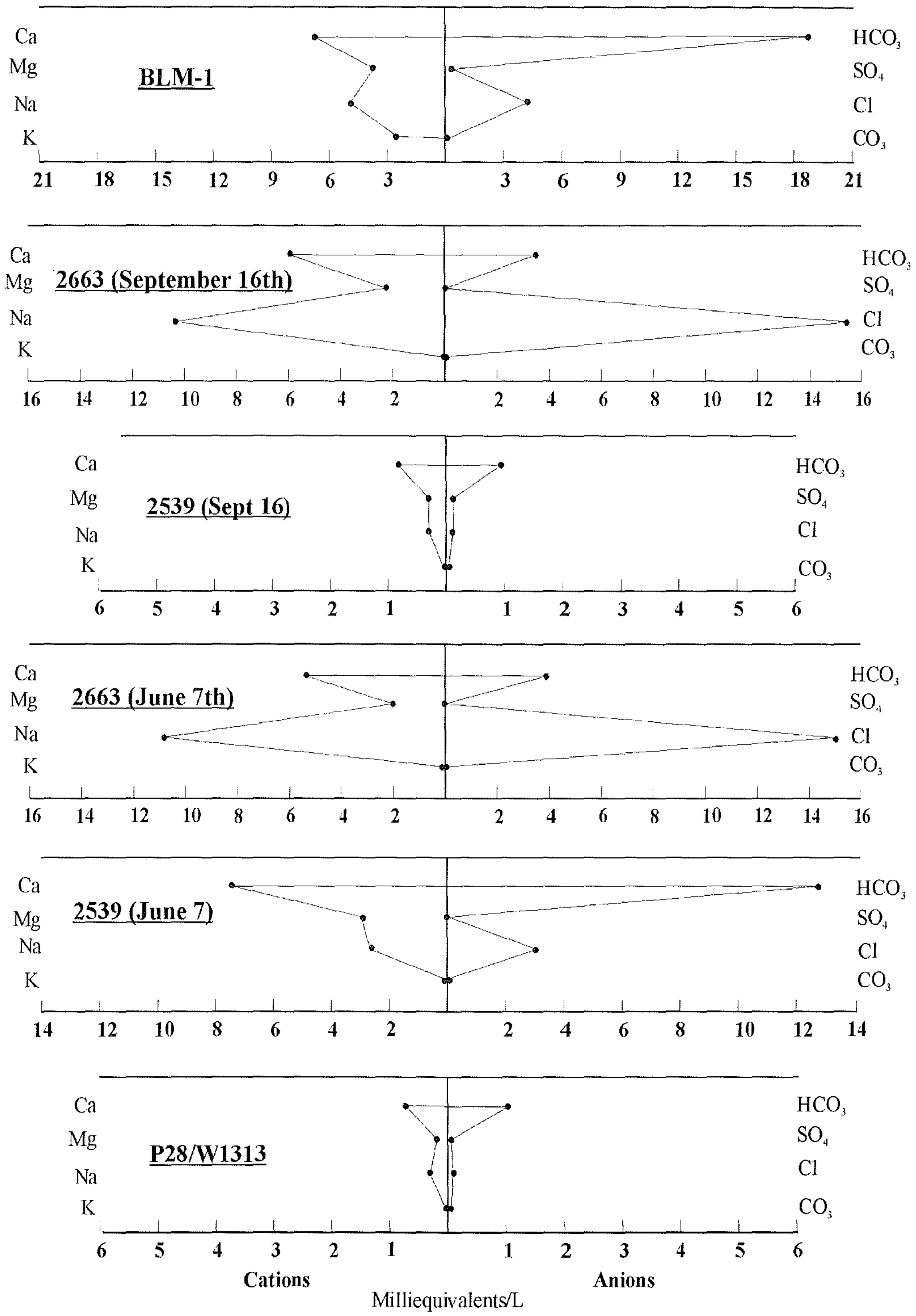


FIGURE 5.8: STIFF PLOT OF MAJOR IONS. A) CONTROL DATA BLM-1 AND P28/W1313, B) AVERAGED FROM SEPTEMBER 1999 AND JANUARY 2000, AND C) P28/W2539 AND P28/W2663 – RESULTS FROM JUNE 2000.

certainly the areas of highest contaminant concentration, is likely to exist below the level tested within P28/W2539. Screening through contaminated and uncontaminated water within and above the contaminant plume probably affects the chemical results from both P28/W2663 and P28/W2539 with mixing of waters diminishing the true concentration of contaminants within the leachate plume. Arsenic is noted in P28/W2663 to be elevated above background levels of 0.06

g/m^3 and from the generally increasing trend, the origin of arsenic in P28/W2663 is interpreted as being landfill leachate. Manganese has not been tested on a long-term basis.

Parameters that were classified as *possible* and *probable* indicators of leachate contamination have been reviewed following analysis of groundwater sampled from immediate down-gradient bores. From Table 5.11, parameters that notably exceed the maximum Taylor Pass background groundwater quality in either P28/W2663 and/or P28/W2539 and that have previously been identified as *possible* leachate indicators can be assumed to be *probable* indicators of leachate contamination. These parameters are:

- sodium,
- chloride, and
- arsenic.

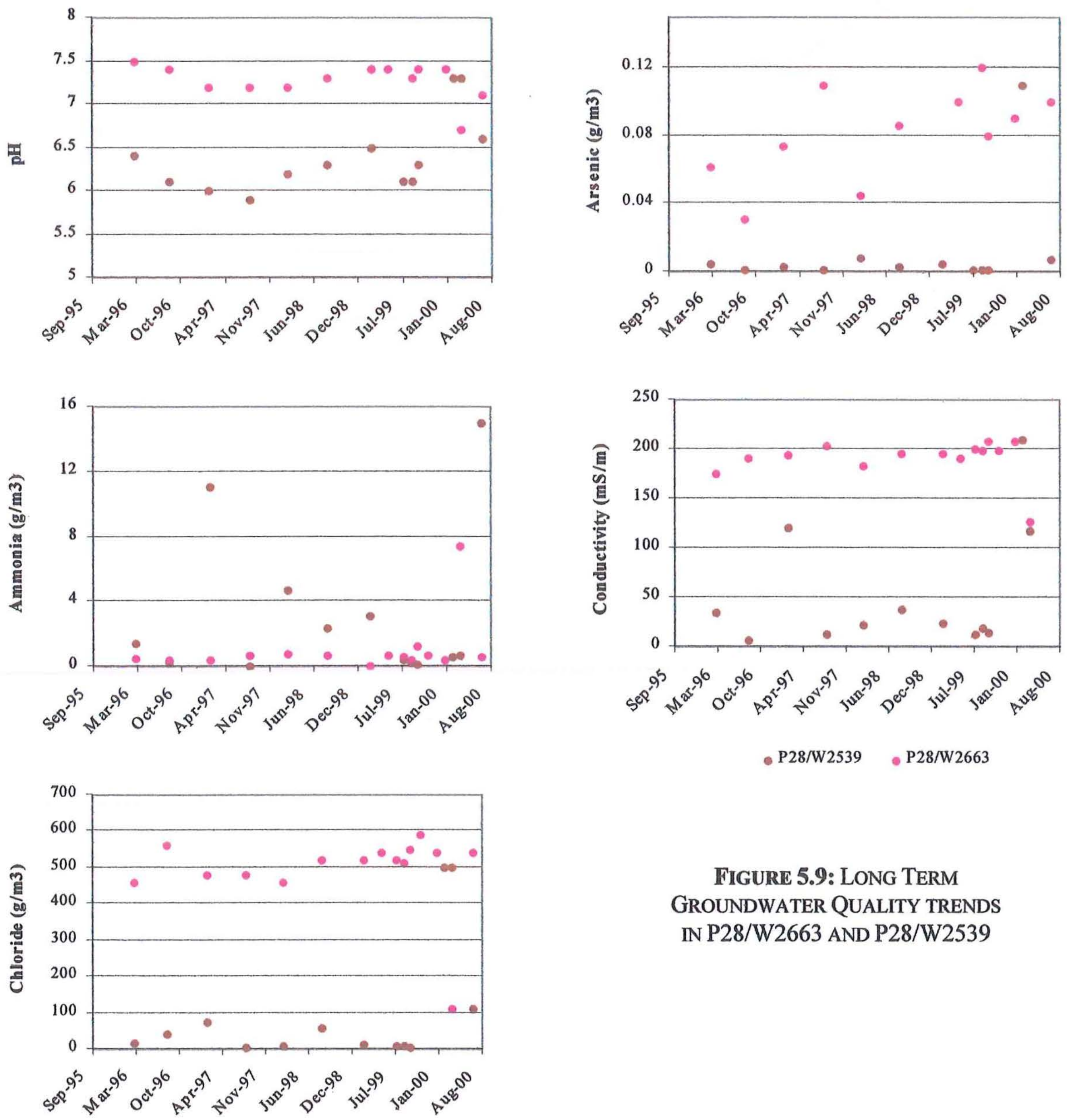


FIGURE 5.9: LONG TERM GROUNDWATER QUALITY TRENDS IN P28/W2663 AND P28/W2539

Hydraulic Relationship – P28/W2663 and P28/W2539

Wells P28/W2663 and P28/W2539 have comparable static water levels, suggesting they are hydraulically connected, but chemically they show very different and often opposing trends as indicated above. The following discussion investigates the deposits that the two wells penetrate and attempts to explain possible mechanisms of groundwater flow that may result in the complex chemical results obtained.

Figure 5.10 shows a south-north cross sectional view through P28/W2663 and P28/W2539 correlating and interpreting well log data for the two bores. P28/W2539 does not penetrate any notable impermeable layers whereas P28/W2663 less than 3 m south of P28/W2539 penetrates and is partially screened through a 10.8 m thick fine gravel and clay layer that is likely to act as an impermeable barrier separating the screened intervals of the two wells.

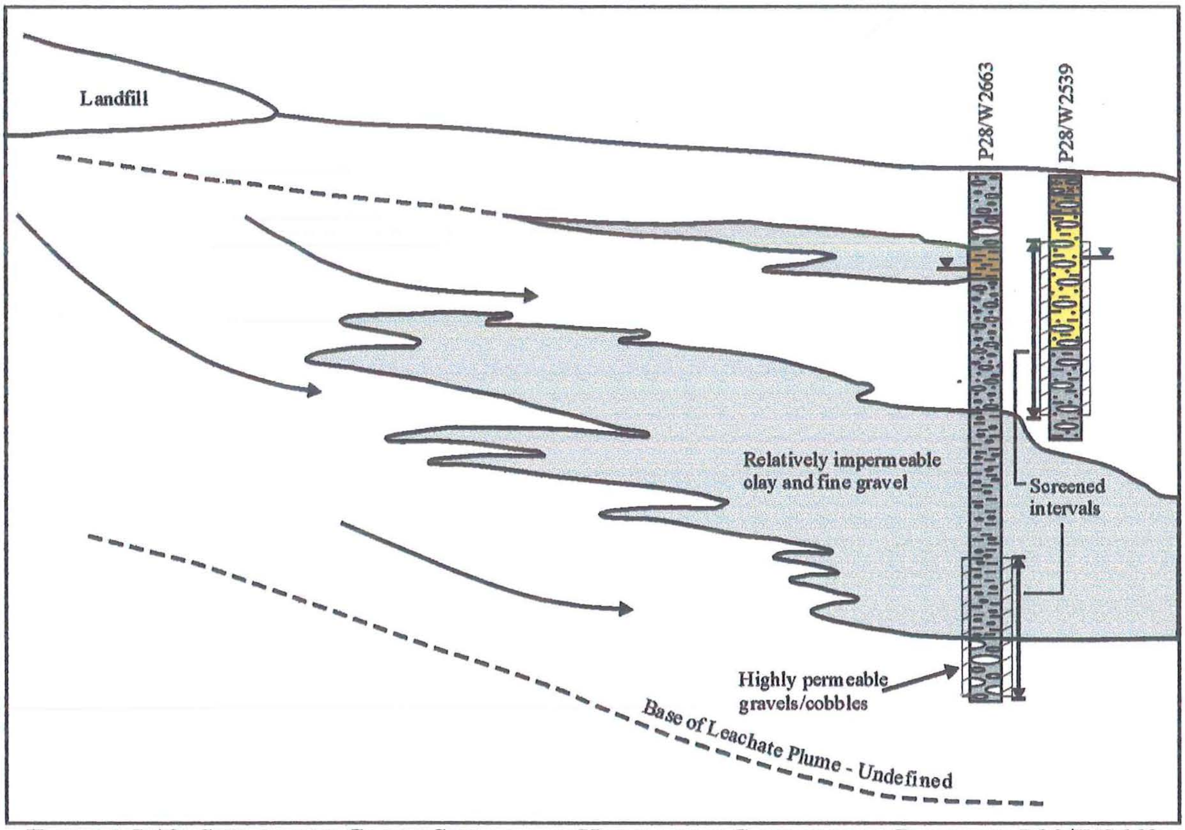


FIGURE 5.10: SCHEMATIC CROSS SECTION OF HYDRAULIC CONNECTION BETWEEN P28/W2663 AND P28/W2539.

The fact that the water quality in the two wells remains significantly different can be explained then by the presence of a dividing impermeable layer between the screened intervals of the wells. The origin of the two different types of waters and the opposing trends of contamination with respect to time is not so easily explained. Over the bottom section of the screened interval of P28/W2663, gravels and cobbles provide a permeable medium through which groundwater flow is likely to be constant and relatively rapid. Contaminants are not likely to be well attenuated in this area, nor are they likely to dramatically change in concentration with fluctuations in the water table. In P28/W2539, however, the gravels have a high clay content, are consequently less permeable and have a high potential for attenuation of contaminants, and the screened interval extends above the water table. Sampled groundwater from P28/W2539 therefore is more susceptible to changes in water quality associated with water tables fluctuations that lead to increased leaching of contaminants from the soil associated with groundwater table rises.

The presence of Wairau-type waters in P28/W2539 is extremely perplexing. The only natural mechanism thought to be able to account the phenomenon is the upwelling of Wairau groundwater at the southern periphery of confining layers associated with aquifers of the Wairau Plains. It is however, unlikely that this would occur so far up the surface of the Taylor Fan, or that upwelling would provide sufficient waters to influence the chemistry of P28/W2539 so markedly while deeper water in P28/W2663 remain so obviously associated with contamination from the Taylor Pass Landfill. Currently there is no feasible explanation for the nature of groundwater in P28/W2539 and further work is recommended to establish its origin. Ascertaining the origin of groundwaters in P28/W2539 would no doubt provide a better understanding of the effects and movement of leachate down-gradient of the landfill.

There is currently little control over the variation in static water levels in the two wells, and further monitoring and investigation of static water level trends correlated with water chemistry may better define the connection between the wells and offer some insight to the obviously complex relationship between P28/2663 and P28/W2539

5.6.4 Other Evidence for Leachate Migration

Introduction

Investigations of leachate migration away from the Taylor Pass Landfill are divided into four broad areas as follows:

- Westward migration (P28/W3388)
- Eastward migration (P28/W3002 and P28/W2618)
- Shallow down-gradient migration (P28/W2619, P28/W2661, P28/W2662 and PP28/W240)
- Deep down-gradient migration (P28/W3389 and P28/W3391)

Depths and details of P28/W1477 at Redwoodtown School and CS-IHC on Cleghorn Street are briefly discussed in conjunction with the deep down-gradient monitoring bores. Investigations of groundwater quality at Aerodrome Road (P28/W3390) are discussed separately in the next section.

Westward Migration – P28/W3388

One hundred metres west of the Taylor Pass Landfill, P28/W3388 samples groundwater from 5-10 m deep in the modern riverbed. The well was installed and monitored primarily to detect any leachate migrating in a westward direction. Results of monitoring are summarised in Table 5.12 and show that none of the tested parameters exceed maximum Taylor Pass background levels in the period between July 1999 and June 2000. Concentrations of major ions (Figure 5.11) indicate that the groundwater in P28/W3388 is typical of Taylor Pass groundwater when compared with background monitoring well P28/W3386. The only contaminant noted in P28/W3388 is manganese with a maximum recorded value of 6.3 g/m^3 in excess of background levels of 2.6 g/m^3 indicating that the well is contaminated with respect to manganese, however the lack of other contaminants and application of the model proposed by Deutsch (1997) suggests that the location of the well represents the maximum westward lateral extent of migration at this point. Any leachate that may be moving in a westward direction must be migrating either deeper than 10 m or passing through the zone between the landfill and the Taylor River further north than P28/W3388. The possibility of westward migration following either or both of these paths is further discussed with reference to the Aerodrome Road monitoring bore (P28/W3390) in Section 5.6.5. With the exception of P28/W3390, no other leachate-monitoring bores exist on the western side of the landfill.

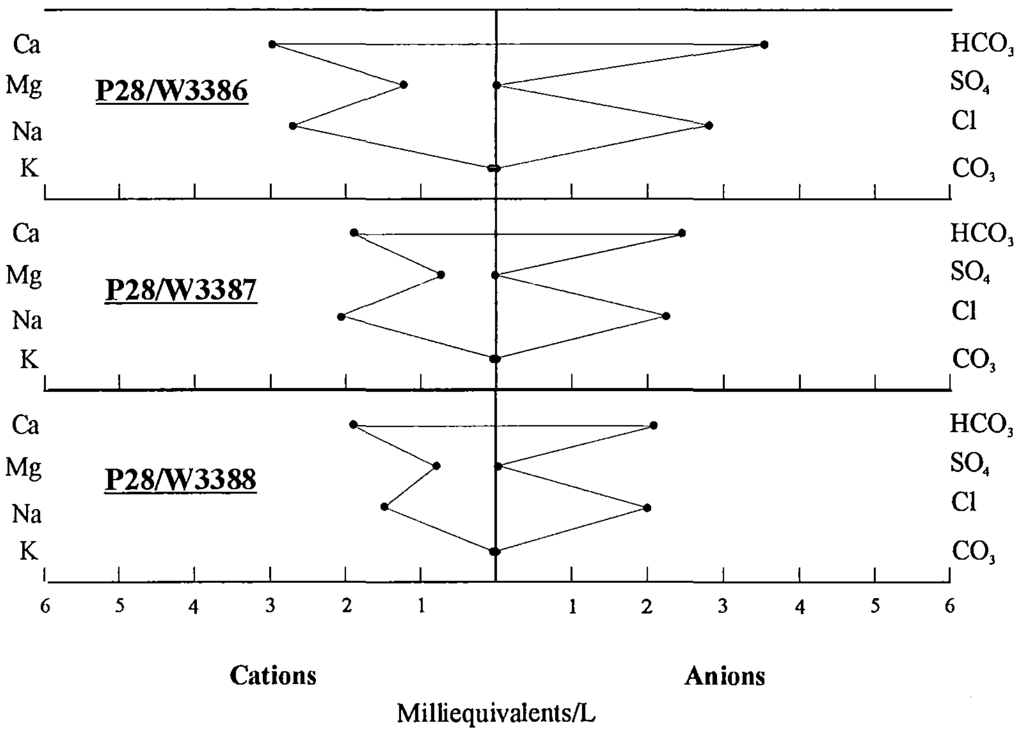


FIGURE 5.11: STIFF PLOTS OF BACKGROUND AND P28/W3388.

Parameter	Units	Maximum Taylor Pass Background Levels	P28/W3388	
			Average	Range
DO	g/m^3	1.04-5.65	4.1	2.18-5.65
Conductivity	mS/m	130	43	40.7-45.3
pH	pH units	7.0-7.5	7.0	7.0-7.1
Alkalinity	g/m^3 as Ca CO_3	250	103	96-110
Bicarbonate	g/m^3	240	130	120-140
Carbonate	g/m^3	<1		<1
COD	g/m^3	21		
Ammonia-N	g/m^3	1.9	0.021	0.01-0.074
Nitrate-N	g/m^3	0.18		
Nitrite-N	g/m^3	<0.001		
Total N	g/m^3	5.2	0.38	0.3-0.46
Total P	g/m^3	2.1	0.66	0.55-0.76
Chloride	g/m^3	310	72	64-78
Sulphate	g/m^3	2.1	1.3	1.2-1.4
Potassium	g/m^3	6.2	1.4	1.4-1.5
Sodium	g/m^3	200	34	29-38
Calcium	g/m^3	78	38	36-41
Magnesium	g/m^3	20	10	9.7-11
Aluminum	g/m^3	2.2	1.80	<0.02-4.8
Arsenic	g/m^3	0.064	0.015	0.013-0.018
Chromium	g/m^3	0.002		
Copper	g/m^3	0.4		
Nickel	g/m^3	0.006		
Lead	g/m^3	0.13	0.02	0.014-0.03
Zinc	g/m^3	0.028		
TOC	g/m^3	16	1.3	0.8-2.4

TABLE 5.12: SUMMARY RESULTS FOR WESTWARD MIGRATION INVESTIGATION – P28/W3388 (MAY 1999-JUNE 2000)

Eastward Migration – P28/W3002 and P28/W2618

P28/W3002 and P28/W2618 are used to assess eastward migration of leachate from the Taylor Pass Landfill (refer Figure 5.2). Results from May 1999 to June 2000 are summarized in Table 5.13. P28/W2618 has only been tested twice over this period as part of the standard Connell Wagner monitoring regime, while P28/W3002 was tested in 7 monitoring rounds over the same period.

Parameters identified at significant above background Taylor Pass levels (or outside of the defined range in the case of pH) are listed below and their relevance to identification of leachate-derived contamination is further discussed below.

- pH (P28/W2618 only)
- COD
- Ammonia-N (P28/W2618 only)
- Sulphate
- Potassium
- Aluminium
- Manganese (P28/W2618 only)
- Total N and Total P

Parameter	units	Maximum Taylor Pass Background Levels	P28/W3002		P28/W2618
			Average	Range	Average
DO	g/m^3	1.04-5.65	5.22	2.39-8.75	
Conductivity	mS/m	130	94	87.4-106.6	136.5
pH	pH units	7.0-7.5	7.3	7.1-7.4	6.55
Alkalinity	g/m^3 as Ca CO_3	250	165	150-180	
Bicarbonate	g/m^3	240	198	190-210	
Carbonate	g/m^3	<1	<1		
COD	g/m^3	21		16*-120*	50.5
Ammonia-N	g/m^3	1.9	0.23	0.13-0.38	20.5
Nitrate-N	g/m^3	0.18			0.11
Nitrite-N	g/m^3	<0.001			
Total N	g/m^3	5.2	3.3	0.68-7.5	
Total P	g/m^3	2.1	2.3	0.95-3.9	
Chloride	g/m^3	310	196	170-240	78
Sulphate	g/m^3	2.1	6.71	<0.005-25	
Potassium	g/m^3	6.2	5	1.9-12	
Sodium	g/m^3	200	123	110-140	
Calcium	g/m^3	78	53	48-58	
Magnesium	g/m^3	20	17	15-19	
Aluminum	g/m^3	2.2	7.65	<0.002-11	
Arsenic	g/m^3	0.064	0.053	0.011-0.087	0.004
Chromium	g/m^3	0.002			0.002
Copper	g/m^3	0.4			
Nickel	g/m^3	0.006			
Lead	g/m^3	0.13	0.084	0.026-0.19	<0.01
Zinc	g/m^3	0.028			
Manganese	g/m^3	2.6	1.5	1.1-2.7	3.8*
TOC	g/m^3	16	3.2	1.8-6.1	

* result from monitoring in 1996 and 1997 - parameter not tested in 1999-2000.

TABLE 5.13: SUMMARY RESULTS FOR EASTWARD MIGRATION INVESTIGATION – P28/W3002 AND P28/W2619.

Elevated total-N and total-P in P28/W3002 of 7.5 and 3.9 g/m^3 in September 1999 cannot be attributed to a landfill source, as it is likely to be affected by fertiliser application in surrounding paddocks approximately two weeks prior to sampling. The application of fertilisers in the immediate vicinity of the landfill and the effect on groundwater noted in the September monitoring round obviously limits the applicability of total-P, total-N, and thus nitrate-N and nitrate-N to investigations of strictly landfill sourced contamination. The parameters are of obvious interest to concerned parties with respect to overall groundwater quality and should be addressed in additional investigations. Due to the ambiguity of the true origin of contamination, further interpretation of total-P, total-N, nitrate-N and nitrite-N levels down-gradient remains beyond the scope of this project.

Aluminium in P28/W3002 of up to 11 g/m^3 is well in excess of any recorded levels in both background groundwater and landfill leachate at the Taylor Pass Landfill. The occurrence of such high Al appears localised and may or may not be at least partially landfill-derived. Colloidal Al from clayey Taylor Fan gravels may also contribute to elevated aluminium.

P28/W3002 displays variable high dissolved oxygen levels between 2.39 and 8.75 ppm. The average DO level of 5.22 ppm is only just below the maximum background level of 5.65 ppm, and pH remains within the background limits. Conductivity, alkalinity, bicarbonate, ammonia-N and chloride, which are all elevated in BLM-1, are all below background levels in P28/W3002, suggesting that the well is not contaminated by landfill leachate. However, sulphate, potassium, arsenic and lead are all variably above background levels in P28/W3002, with maximum values of 25, 12, 0.087, and 0.19 g/m^3. These parameters are all likely to be landfill sourced and indicative of intermittent mixing of leachate and Taylor Pass background waters east of the landfill.

Contamination in P28/W2618, located on the immediate eastern perimeter of the Taylor Pass Landfill, is typical of that expected from a landfill source. COD is elevated above background to 50.5 g/m^3 and pH is slightly acidic (6.6). Slightly acidic pH values are noted in Wairau groundwaters also (6.5-6.9 in P28/W1313) but given the location of P28/W2618 on the perimeter of the landfill and that acid pH is typical of landfill leachate, acid pH is in P28/W2618 interpreted as being primarily due to leachate contamination. Conductivity is relatively low (maximum of 136) given the proximity of the sampling site to the landfill. Ammonia-N values of 20.5 g/m^3 corresponding to a 1000% increase above background are obviously landfill sourced, yet manganese levels tested twice in 1996 and 1997 are only 46% higher than maximum background.

Organic Contaminants

Connell Wagner carries out testing a range of organic compounds on a regular basis at P28/W2618 and on volatile organic compounds in other standard monitoring regime wells. Full results are tabulated in Appendix 5, and are summarised as follows:

- Volatile organic compounds comprising benzene, toluene, ethyl benzene and xylenes have been tested on a six monthly basis with varying results. The presence of these compounds indicates the possibility of a light non-aqueous phase liquid plume migrating from the eastern periphery of the Taylor Pass Landfill. The compounds are evident in P28/W2619 in more dilute concentrations indicating down-gradient attenuation probably by both dispersion and sorption. The compounds are not detected further down-gradient.
- Tested polynuclear aromatic hydrocarbons (PAHs) tested in P28/Wshow a decrease of between 4-60% from March 1999 to June 2000 however this is based on only two tests and may not be representative of the true change in concentrations. No background levels have been established and the nature of PAHs in a leachate plume is difficult to adequately characterise based on only two monitoring rounds.
- A selection of other volatile halogenated organic compounds (refer Appendix 5) have been tested once in P28/W2618 and although their presence is noted, the lack of background

control and a shortage of comparative down-gradient data limits the usefulness of analyses for the delineation of a leachate plume. Naphthalene was tested in both P28/W2618 and P28/W2619 in 1996 indicating and down-gradient decline in concentrations from 570 to 223 mg/m^3. No drinking water guideline value exists for naphthalene.

Shallow Migration Monitoring – Additional Connell Wagner Monitoring Bores

Wells classified as shallow down-gradient monitoring wells are P28/W2619, P28/W2662, P28/W2661 and P28/W2540. All the shallow monitoring wells are screened above 11.8 m except P28/W2619, which only penetrates to 11.2 m. The wells form a line parallel to Taylor Pass Road down-gradient of the Taylor Pass Landfill, with P28/W2540 farthest away on the southern side of New Renwick Road. P28/W2540 and P28/W2661 are located adjacent and immediately down gradient of the former Brayshaw Park Landfill respectively (refer Figure 5.2), and should exhibit effects of any leachate generated from this source.

Monitoring of the wells by Connell Wagner has been undertaken on a six monthly basis with 9 monitoring rounds completed from March 1996 to February 2000. Selected parameters are displayed graphically in Figure 5.12 to indicate the change in concentration of leachate indicators with time. P28/W2539 (discussed previously) is also a shallow well and is included for comparative purposes. Although caution has been advised with regard to boron as a leachate indicator, boron results have been included in Figure 5.12 as they appear to show similar trends to arsenic, which has been defined as indicative of leachate based on results from immediate down-gradient bores.

Results show highly variable levels of all major leachate indicative parameters tested regularly in shallow monitoring wells over the period from March 1996. Conductivity of waters in P28/2619, P28/W2662 and P28/W2661 varies in the vicinity of the upper limit for background Taylor Pass groundwaters of 130 mS/m, and no consistent down-gradient increase or decrease can be determined. Therefore, conductivity is not considered a reliable indicator of shallow leachate contamination down-gradient of the Taylor Pass Landfill. The implications of the lack of distinctive leachate- conductivity are discussed further in Section 5.7.1 with reference to geophysical investigations.

An ammonia phase within the leachate plume appears at levels of up to 24 g/m^3 in P28/W2619 in August 1998, before diminishing to a low of 4.8 g/m^3 in February 2000. This trend is reflected in COD levels in P28/W2619, and the source of contamination is interpreted as being Taylor Pass Landfill leachate-derived. Chloride, arsenic, conductivity and boron values are all below background levels, and with the exception of conductivity, are all significantly less than in the next down-gradient well P28/W2662.

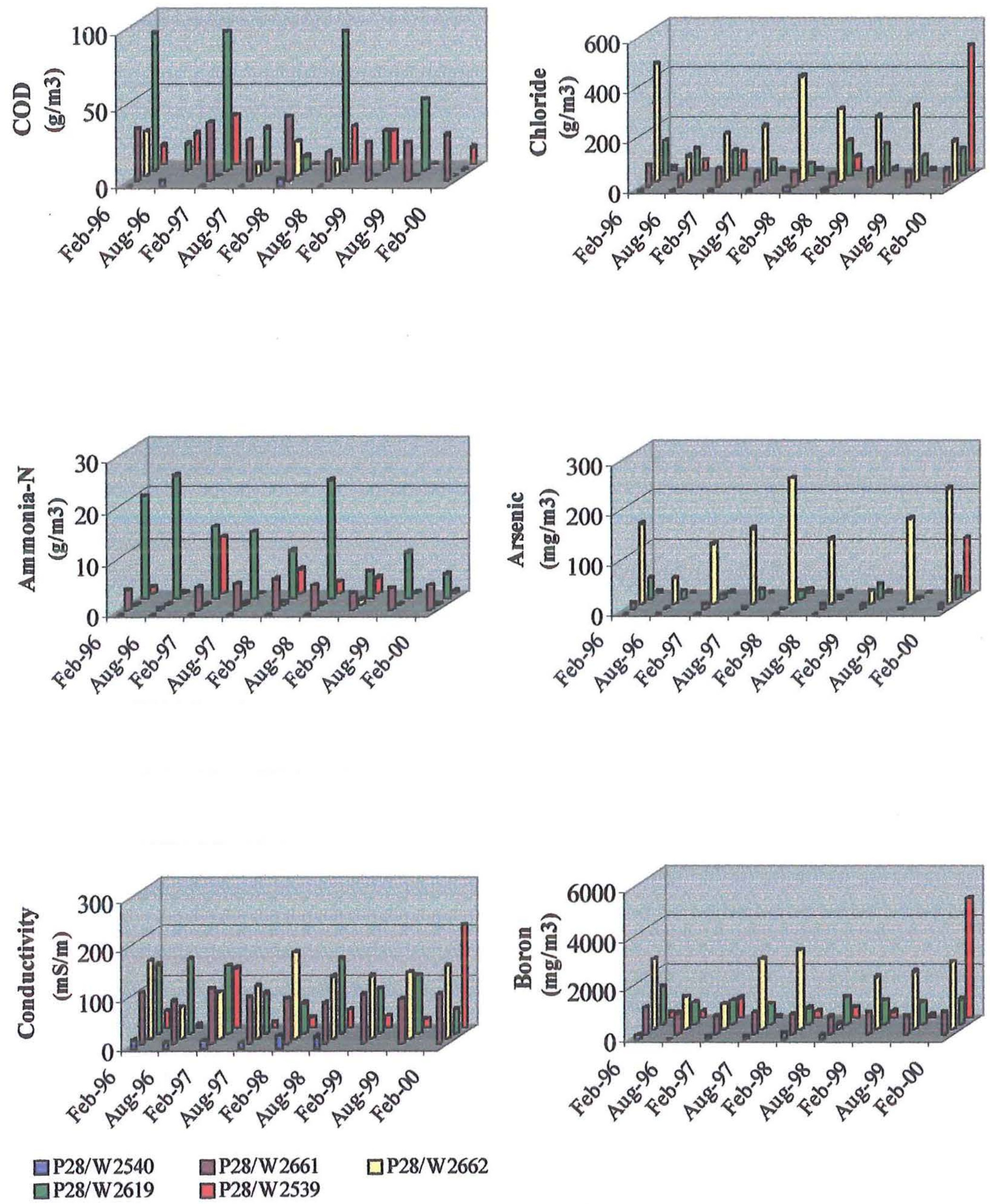


FIGURE 5.12: SUMMARY GRAPHS OF SHALLOW GROUNDWATER QUALITY MONITORING. WELLS ARRANGED FROM P28/W2540 AT FRONT OF GRAPH TO CONTROL WELL P28/W2539 AT REAR.
REFER TO LEGEND ABOVE FOR COLOUR CODING

Chloride and arsenic levels in P28/W2662 are both variably above background Taylor Pass concentrations, with maximum values of 470 g/m^3 and 0.25 g/m^3 respectively. This may indicate a slightly faster moving phase of the leachate plume in which ammonia-N is absent due to retardation in clayey gravels, but the appearance of ammonia again in P28/W2661 implies that some currently undefined mechanism is acting to reduce ammonia levels in P28/W2662. Although P28/W2661 is adjacent to and down-gradient of the Brayshaw Park Landfill, it is unlikely that ammonia levels at this location can be attributed to leachate from Brayshaw Park given that the site has been closed for 25 years and is not likely to be still actively generating significant ammonia.

Manganese, which has been identified as a leachate indicator, has not been tested regularly in the shallow monitoring wells down-gradient of the Taylor Pass Landfill. The parameter was tested in March 1996 and February 1997 with results as follows:

- P28/W2619 – 2.4 g/m^3 (both monitoring rounds)
- P28/W2662 – 4.3 and 0.69 g/m^3
- P28/W2661 – 4.0 and 3.2 g/m^3
- P28/W2540 – 0.0013 and < 0.01 g/m^3

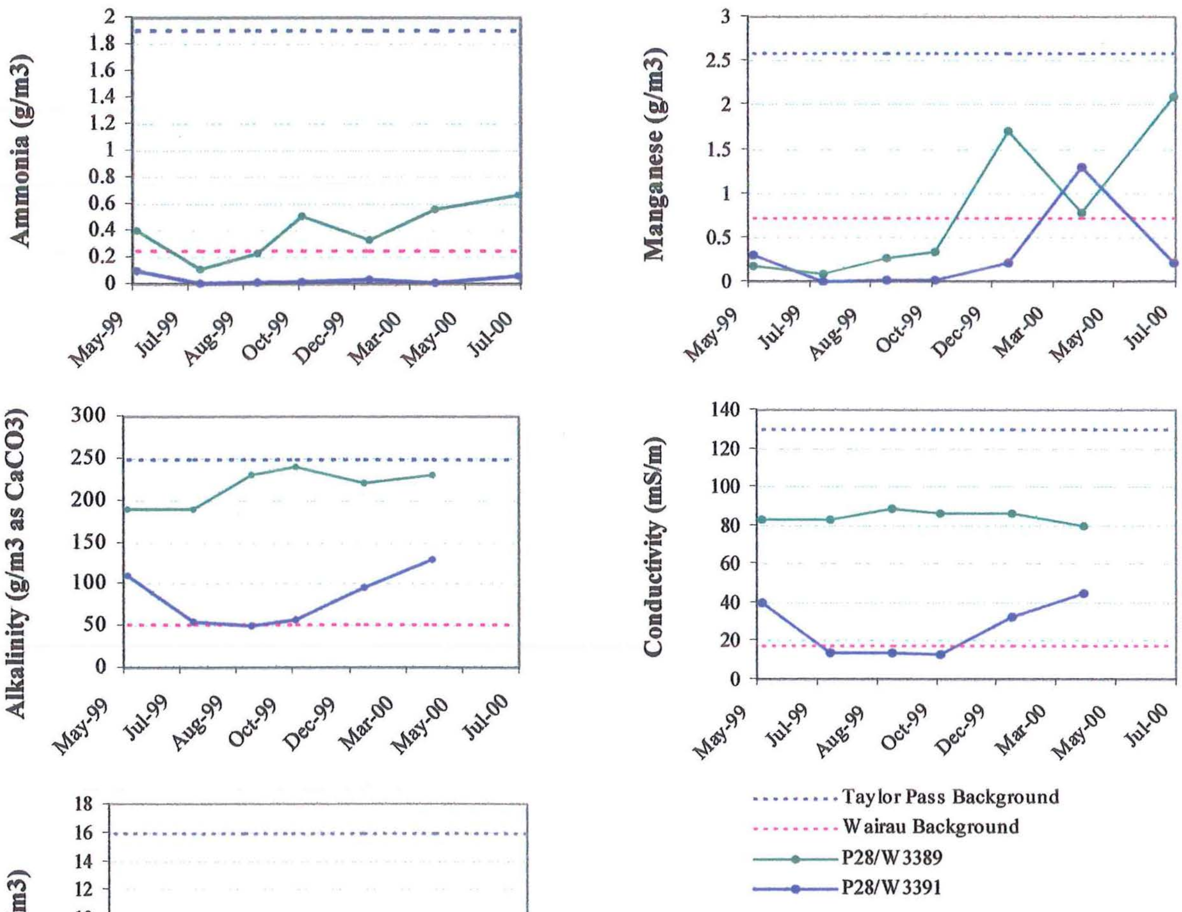
Manganese levels recorded in P28/2661 in excess of maximum background groundwater levels of 2.6 g/m^3 suggests that groundwater may be contaminated by leachate to at least 800 m down-gradient of the Taylor Pass Landfill. It must be acknowledged however that elevated manganese cannot be relied upon solely as a leachate indicator at P28/W2661 as levels are only 1.4 times that of background ground water and could represent an extraordinary natural concentration.

P28/W2540 at the northern end of Brayshaw Park Landfill had been tested up until 1998, with results indicating low levels of all of major leachate indicators. Maximum COD levels of 5 g/m^3 have been recorded but are generally undetected. Chloride has been detected to 19 g/m^3 in comparison to background Taylor Pass levels of up to 200 g/m^3, and Wairau groundwater control levels of 9 g/m^3. Arsenic remains undetected, whilst ammonia values have been reported to 0.012 g/m^3. Conductivity is up to two times higher than Wairau control values, ranging from 13-27 mS/m, but in P28/W2540 remains well below Taylor Pass background conductivity. Results from P28/W2540 indicate that the tested water cannot be considered contaminated from either the Taylor Pass or Brayshaw Park landfills. Any shallow leachate from the Taylor Pass Landfill that may have migrated up to 1 km down-gradient should be detected in P28/W2540. In the absence of any notably high leachate indicator parameters in P28/W2540, it must be assumed that shallow leachate from the Taylor Pass landfill either has not migrated thus far down the Taylor Fan or is sufficiently diluted and dissipated along its migration path down the Taylor Fan that when mixing of Taylor Fan and Wairau waters occurs in the vicinity of New Renwick Road, any remaining contaminants at high concentrations are further diluted to levels where they cannot be distinguished from natural background levels. Shallow leachate from Brayshaw Park, which is likely to be being generated at significantly lower rates and toxicity than the Taylor Pass Landfill, should also be present in P28/W2540. The lack of evidence of contaminants indicates that either leachate is no longer being actively generated at the Brayshaw Park site, or the mixing of Taylor Pass and Wairau groundwaters effectively reduces the concentration of contaminants and masks the effects of any shallow leachate migration north of New Renwick Road.

Deep Migration Monitoring – P28/W3389 and P28/W3391

Migration of leachate down-gradient of the Taylor Pass Landfill at depths greater than 15 m is monitored at newly installed wells P28/W3391on New Renwick Road, and at P28/W3389 on Page Street (refer Figure 5.2). Both wells were installed as a part of this project and have been monitored in 6 project-specific monitoring rounds to March 2000 and one additional monitoring round in June 2000.

P28/W3391 is screened from 27-32 m below the ground surface, and samples water from deeper than any other monitoring bores used as part of either the standard Connell Wagner network or the specific monitoring regime designed for this project. P28/W3389 on Page Street is screened between 20.8 and 25.6 m. Selected parameters are plotted in Figure 5.13.



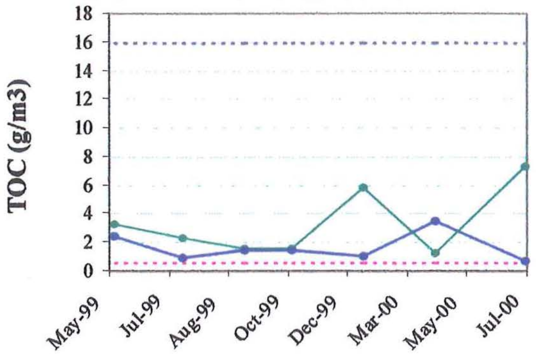


FIGURE 5.13: SUMMARY GRAPHS OF DEEP GROUNDWATER QUALITY AND COMPARISON WITH MAXIMUM BACKGROUND TAYLOR PASS AND WAIRAU GROUNDWATER FROM SAMPLING OVER THE PERIOD MAY 1999 TO (29) JUNE 2000.

Results indicate that major leachate indicator parameters of ammonia, manganese, alkalinity and TOC in P28/W3391 are all present in concentrations of the same order as maximum Wairau levels. There is no indication of a rising trend in any parameters in P28/W3391, and conductivity and ammonia and manganese appear to follow the same trend of low levels in July through to August followed by an increase to May which, by the smooth pattern of change and its apparent cyclic nature, may be interpreted as a seasonal fluctuation. As no obvious increasing trend exists in data from P28/W3391 it can be assumed that the elevation above background Wairau levels is a function of mixing of Taylor Pass and Wairau groundwaters without any significant leachate contamination.

Concentrations of parameters indicating leachate contamination in P28/W3389 (Page Street) are all in excess of levels in P28/W3391, reflecting its location approximately 300 m south of New Renwick Road within true Taylor Fan deposits. The groundwaters at this location are obviously still being significantly affected by Wairau groundwater, as indicated by parameter concentrations generally well below background Taylor Pass levels. An overall increasing trend in ammonia, manganese, alkalinity, and to a lesser extent TOC, from May 1999 to June 2000 suggests that the well is displaying initial signs of contamination by a likely landfill-derived source. Ammonia concentrations should be significantly diminished due to retardation along its migration path however, there is little control over its movement from the landfill and thus the effects of retardation are unable to be determined. The lack of an increasing trend in conductivity reflects the fact that the conductivity of the leachate plume is significantly masked by the conductivity of background Taylor Pass groundwater, and cannot be reliably interpreted as indicative of contaminated or uncontaminated groundwater at significant distances from the landfill. The down-gradient extent to which conductivity can be regarded as indicative of leachate contamination is currently undetermined.

Other Monitoring Bores

Regular landfill monitoring carried out by Connell Wagner Ltd includes testing of shallow groundwaters (<10 m) from P28/W1477 and CS-IHC in the Redwoodtown area. Full results of testing are tabulated in Appendix 6 and are summarised as follows:

- Conductivity values range from 33-43 and11-54 mS/m in P28/W1477 respectively
- COD is of the order of 5-8 g/m^3 in both wells, however in August 1996 and September 1997 levels of 15 and 18 g/m^3 were recorded in P28/W1477. This is not considered significant, as levels have remained steady at 6-8 g/m^3 since February 1998.

- Ammonia-N remains largely undetected in both wells, with maximum values of 0.019 g/m^3 in CS-IHC. Chloride remains below 35 g/m^3 in both wells throughout all monitoring rounds.

Parameters tested in groundwaters sampled from CS-IHC and P28/W1477 are comparable with Wairau-derived ground water. Chloride and consequently conductivity levels slightly higher than Wairau derived water are interpreted as being due to mixing of Taylor Pass and Wairau derived groundwaters. The Taylor Pass Landfill does not contaminate Wells P28/W1477 and CS-IHC.

5.6.5 Aerodrome Road Bore – P28/W3390

Background

P28/W3390 is located at the corner of Aerodrome Road immediately northeast of the Omaka Aerodrome (refer Figure 5.2) within Speargrass Formation Taylor Fan gravels near the boundary of the modern Rapaura Formation Taylor Fan lobe. The well is screened from 13.8-18.8 m below ground surface and has been tested during six project specific monitoring rounds from May 1999 to March 2000, and an additional monitoring round in June 2000. Interpretation and discussion of monitoring results from P28/W3390 have been isolated from other monitoring sites due to both the enigmatic nature of results and the close down-gradient proximity of the well to the Omaka Aerodrome, which represents another potential contamination source. Water quality at the monitoring well is introduced, followed by a discussion of the possible origins of contamination.

Composition

The quality and variability of water quality at P28/W3390 is characterised by the following attributes:

- Steadily decreasing pH from 12-7.2 over a period from May 1999 to June 2000. Low acidity tested once in 1999 indicates the poor base-neutralizing capacity of the system.
- Overall low dissolved oxygen ranging from 0.066 to 1.7 ppm.
- Steady bicarbonate of 200-220 g/m^3 in monitoring rounds in 2000, following high levels in May 1999 (1100 g/m^3) and contrasting low levels in September 1999 (52 g/m^3). Alkalinity is dominated by the bicarbonate ion.
- Carbonate decreasing from 420 g/m^3 in May 1999 to <1 in June 2000, and calcium and potassium also steadily decreasing from 330 to 36 g/m^3 and 16 to 1.1 g/m^3 respectively.
- Ammonia-N fluctuates between 1.3 and 0.56 g/m^3 between July 1999 and June 2000. Total-N in May 1999 is recorded as 2.1 g/m^3, which directly contradicts ammonia-N levels

of 3.7 g/m^3 in the same monitoring round. May 1999 results for both parameters must therefore, be disregarded.

- Constant sodium levels of 170 g/m^3 are comparable with levels of greater than 230 g/m^3 recorded in P28/W2663, and a one-off sample from BLM-1 in March 1996 of 180 g/m^3. Chloride is consistently between 200 and 230 g/m^3.
- Total phosphorus significantly decreases from 1.7 g/m^3 in May 1999 to 0.078 g/m^3 in July 1999, then recovers to 1.1 g/m^3 in January 2000. Phosphorus may originate from up-gradient fertiliser application and/or fertiliser residue from topdressing aircraft using the up-gradient airstrip. Sulphate opposes the phosphorus trend, with levels of up to 13 g/m^3 in July 1999, decreasing to undetectable levels in June 2000.
- TOC is generally between 1 and 4 g/m^3 with levels rising to 21 and 38 g/m^3 in May 1999 and September 1999. Arsenic increases from 0.011 to 0.099 g/m^3 between May 1999 to June 2000.
- Relatively consistent boron of 1.3-1.6 g/m^3, except for extremely low levels in May 1999 of 0.06 g/m^3. Highly variable iron and manganese of 0.12-53 g/m^3 and <0.03.2 g/m^3 respectively. Aluminium is also highly variable between 0.04 and 5.4 g/m^3.
- Conductivity does not obviously reflect trends of any single parameter, with a decreasing trend from May 1999 to November 1999 and increasing again through to March 2000. Conductivity ranges from 97.5 to 467 mS/m.

Discussion of Aerodrome Road Monitoring Results

P28/W3390 is variably contaminated above maximum background levels by:

- undefined organic compounds reflected by variable high TOC
- calcium
- manganese
- iron
- arsenic
- aluminium
- bicarbonate
- sulphate
- potassium

As previously discussed, naturally occurring concentrations of iron are highly variable in both Taylor Pass and Wairau groundwaters, and cannot be reliably used as contamination indicators.

The presence of calcium and an abnormally alkaline pH of up to 12 suggest groundwaters in the Aerodrome Road well originate from an alternative source to groundwaters tested on the eastern side of the Taylor River. Elevated results of these parameters at Aerodrome Road do, however, decrease with time to levels comparable with other monitoring wells on the eastern side of the

Taylor River. Ammonia-N, which is an obvious indicator of contamination in shallow wells on the eastern side of the Taylor River, remains below maximum background levels in P28/W3390 with the exception of anomalous levels in May 1999. The lack of ammonia-N suggests that contamination of the Aerodrome Road bore may not in fact originate from a landfill source.

Background levels of sodium have been noted in BGBH-1 to 200 g/m^3 however P28/W3386 and P28/W3387 show consistent sodium levels between 45 and 70 g/m^3. In P28/W3390, sodium levels of 170 g/m^3 may or may not be associated with contamination from a landfill source. TOC trends suggest contamination by organic compounds occurs in pulses rather than as consistent flow. This trend is analogous to contamination on the eastern side of the Taylor River, however distinct pulses of leachate released from the landfill are likely to be diluted with increasing distance from the landfill. Concentrations of TOC or any other contaminant from the Taylor Pass Landfill are unlikely to be detected at Aerodrome Road, some 800 m from the landfill, in concentrations equivalent or even comparable to concentrations detected in the immediate vicinity of the landfill. It therefore appears likely that TOC contamination of P28/W3390 is not landfill-sourced, and that some other source of contamination must exist in the area.

The Australian and New Zealand Environment and Conservation Council (1992) identify airports as potentially contaminated sites with respect to groundwater. The Omaka Aerodrome is used and has been used mainly by topdressing and recreational aircraft, and was in use during World War II. By the very nature of activity at the site it is possible that TOC, Ca and CO_3 contamination of P28/W3390 may be attributable to aircraft fuels and lime respectively.

Due to the proximity of P28/W3390 to the Omaka Aerodrome, any contamination of the well cannot immediately be interpreted as landfill-derived and given the location of P28/W3390 within the older Speargrass Formation gravels, the prospect of migration via an old buried channel to P28/W3390 is not considered to be hydrogeologically feasible. Evidence to date is that migration is occurring via Rapaura channels oriented NNE from the Taylor Pass Landfill and that little or no westward movement is taking place.

The true source of contamination of P28/W3390 currently remains undetermined. Further investigative work in the area west of the Taylor River is highly recommended to help clarify contamination sources and likely migration paths. It is also suggested that to assist in determining the true extent and source of contamination in the Aerodrome area, a further background monitoring bore be installed within the Speargrass lobe of the Taylor Fan at least 500 m upgradient of the southern boundary of the Omaka Aerodrome.

5.7 Plume Delineation – Analysis and Synthesis

5.7.1 Methodology

In locating a leachate plume, one must incorporate a number of methods and ideas, and make a number of assumptions about the nature of the environment in question. Methods most commonly used in the identification of the extant of a leachate plume are water chemistry analysis and geophysical investigations, with hydrogeological parameters acting to constraining tool to identify the minimum and maximum potential extent of migration. All three investigative tools have been used at the Taylor Pass Landfill and are discussed in following sections.

5.7.2 Geophysical Investigations

Geophysical investigations are used often for the delineation of leachate plumes due to the contrast in electrical properties that typically exists between natural, uncontaminated water and the target leachate (e.g. Greenhouse and Harris, 1983; Armstrong, 1993; Cardarelli and Bernabini, 1997). In the Taylor Pass area, however, the contrast in conductivity between natural background and contaminated waters has been shown in hydrogeochemical investigations to be significant only in the area immediately down-gradient of the landfill. Also, TEM surveys (Chapter 3) have shown that the quality of data obtained down-gradient of the Taylor Pass Landfill is greatly diminished, primarily due to the effects of anthropogenic noise. In an attempt to overcome these problems, the resistivity method (refer Chapter 3 and Appendix 3) was selected as the technique likely to be least affected by anthropogenic noise, and trial geophysical investigations in the form of resistivity profiles were conducted immediately down-gradient of the Taylor Pass Landfill where it was expected that any effects of leachate would be most pronounced and hence distinguishable from background groundwater.

A geoelectric cross section of the raw data and smooth models obtained is shown in Figure 5.14, and the effects of noise are immediately obvious in the raw data displayed as resistivity curves. Data at virtually all stations are poor quality indicated by often random scattering of points and a lack of similarity between stations which are only 20 m apart. On the basis of poor data quality at all of the sites studied, geophysical investigations are deemed to be of little use in the identification of leachate migration from the Taylor Pass Landfill and hence have not been further used in the identification of a leachate plume.

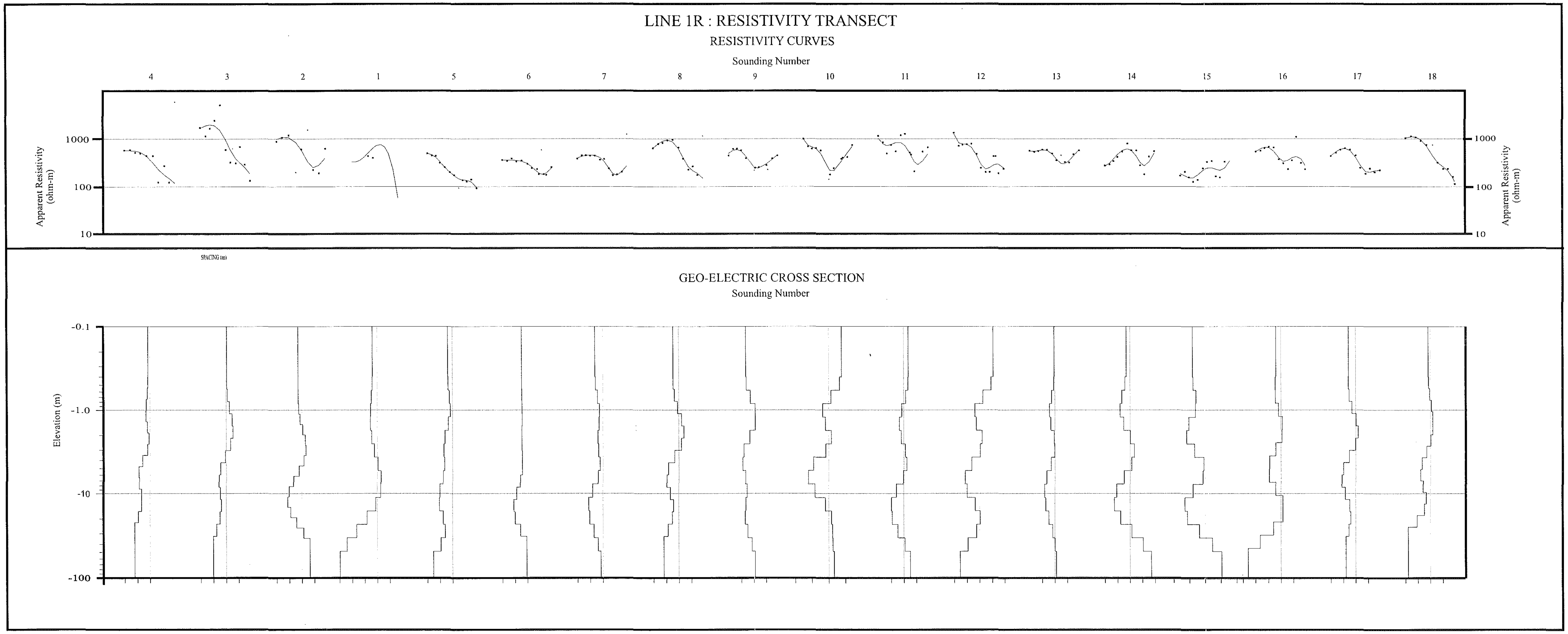


FIGURE 5.14: SAMPLE DATA FORM RESISTIVITY PROFILE DOWN-GRADIENT OF TAYLOR PASS LANDFILL. NOTICE LACK OF QUALITY DATA. SECTION LOOKING NORTHWARDS.

5.7.3 Results from Hydrogeochemical Indicators and Hydrogeological Constraints

Taylor Pass Landfill Leachate Migration

Given the poor results of geophysical investigations, delineation of a leachate plume from the Taylor Pass Landfill is based primarily on the results of the hydrogeochemical investigations discussed above, and is clearly constrained by groundwater seepage velocities assumed for the Rapaura Lobe of the Taylor Fan (refer section 4.4.4).

The interpreted plume extent based on hydrogeochemical and hydrogeological investigations to date is given in Figure 5.15 Parameters that have been used to define the extent of the plume are distinguishable to various distances from the landfill based on their mobility and the amount of masking by naturally high background levels on Taylor Fan. Parameters used are:

- ammonia-N
- TOC
- sodium
- bicarbonate
- chloride
- arsenic
- sulphate
- alkalinity
- manganese
- potassium
- COD
- conductivity

Groundwater chemistry analysis indicates that the groundwater is likely to be variably contaminated to at least 800 m down-gradient of the Taylor Pass Landfill (i.e. appearing in the Page Street well). High background salinity associated with high evapotranspiration and low rainfall however, act to mask the effects of the plume and make distinction between contaminated and uncontaminated water difficult.

Control on migration is provided by P28/W3388, which is only elevated in Mn and suggests that further westward migration is unlikely to be significant, this cannot be entirely ruled out, however, due to a) uncertainty regarding the origin of contaminants in P28/W3390, and b) the piezometric surface in Figure 4.10 which indicates that P28/W3390 must indeed be considered down-gradient of the Taylor Pass Landfill. The difference in overall chemistry in P28/W3390 compared to either background Taylor Pass waters or groundwater down-gradient of the Taylor Pass Landfill (i.e. alkaline pH, presence of calcium and low sodium) strongly suggests however that contamination is not sourced from the Taylor Pass Landfill. Hydrogeologically, it is most unlikely that leachate would migrate across the Taylor Fan when preferred high transmissivity channels are likely to follow the NNE trend of the modern Rapaura lobe.

Eastward of the landfill, the effects of the leachate plume are seen in P28/W3002 ("Gas Bore"), and more strongly in P28/W2618 on the immediate periphery of the Taylor Pass Landfill. A hydrocarbon phase is evident migrating from the eastern periphery of the landfill and has been detected to approximately 100 m down-gradient of the landfill, in shallow wells suggesting a slow-moving LNAPL plume.

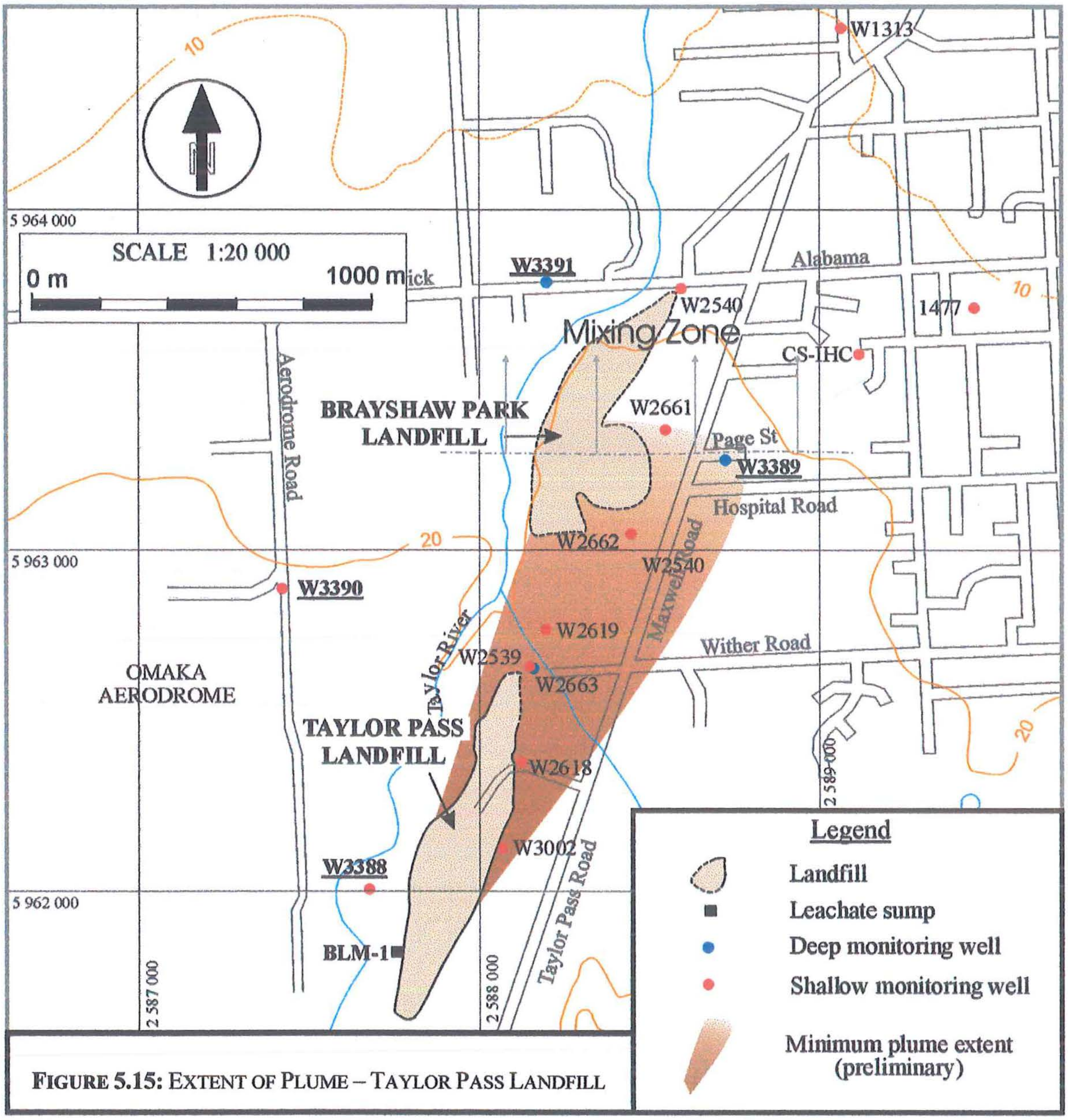


FIGURE 5.15: EXTENT OF PLUME – TAYLOR PASS LANDFILL

Shallow and deep wells indicate different patterns of migration, and adjacent deep and shallow wells P28/W2663 and P28/W2539 often oppose each other indicating a complex vertical hydraulic relationship between layers in the heterogeneous Taylor Fan deposits. There is currently no control over the maximum vertical extent of the leachate plume from the Taylor Pass Landfill, but leachate is detected in P28/W2663 immediately down –gradient of the landfill over a screened interval from 19.0 to 24.8 m. Given the nature of the screened interval comprising permeable cobbles and gravels overlain by likely impermeable clay-bound fine gravels it is expected that sampled water comes from the lower portion of the screened interval thus the screen base of 24.8 m is a true representation of the minimum depth of the leachate plume, and P28/W2663 is not considered to penetrate through to uncontaminated waters below the zone of leachate contamination.

Down-gradient of the Taylor Pass Landfill leachate is distinguished in shallow wells by elevated chloride, COD, ammonia and arsenic levels. At approximately the same distance from the landfill, and indicating the maximum recognisable extent of the leachate plume in deeper groundwaters, contamination of P28/W3389 (Page Street) is characterised by ammonia, alkalinity and manganese. This is interpreted to indicate the approximate down-gradient extent of leachate currently being derived from the most recent refuse dumped in the southern part of the Taylor Pass Landfill

Assuming leachate generation began immediately following the commencement of landfilling operations at Taylor Pass Landfill in 1976, and given minimum estimated seepage velocities of 0.5 m/day in Rapaura gravels of the Taylor Fan surface (Chapter 4), the theoretical potential minimum extent of leachate migration is upwards of 4 km. Therefore, even the theoretical minimum extent of leachate would suggest that the leachate plume has reached and passed the boundary between the Taylor and Wairau Aquifers only 1.5 km north of the landfill. At a minimum seepage velocity of 0.5 m/day it would in fact only take eight years for leachate to migrate from the Taylor Pass Landfill to New Renwick Road and consequently mix with Wairau waters. Therefore it appears likely that the absence of contaminants in wells in the vicinity of New Renwick Road is not necessary because the leachate plume has not extended this far, but rather that the leachate plume, once it reaches the Wairau aquifer, is effectively diluted and dispersed upon mixing of Taylor Pass and Wairau groundwaters.

Influence of the Former Brayshaw Park Landfill

It was expected that contamination by leachate emanating from the Taylor Pass Landfill would (if present) be seen in wells P28/W2540 and/or P28/W3391 representing shallow and deep monitoring bores respectively. P28/W3391 shows no significant differences from Wairau waters as defined by P28/W1313, and it is not considered to be contaminated by leachate. Alkalinity and consequently conductivity values above maximum Wairau groundwater concentrations are interpreted as being a result of mixing of Taylor Pass and Wairau groundwaters, rather than dilute leachate from the Brayshaw Park Landfill. P28/W2540 also show anion and cation concentrations of the order of Wairau-derived groundwater.

In the absence of leachate contamination in wells P28/W3391 and P28/W2540, it is concluded that either a) refuse disposed at Brayshaw Park is sufficiently stabilised that no further leachate is being actively produced, or b) that any leachate being produced, which will be of significantly lower toxicity than that being actively produced at the Taylor Pass Landfill, is being significantly diluted by Wairau groundwater that its effects become indistinguishable in the environment of mixing between Taylor Pass and Wairau derived groundwaters.

5.7.4 Concluding Remarks

Despite hydro-chemical evidence that the present leachate plume extends only about 1.5 km down-gradient from the Taylor Pass Landfill, it must be concluded that earlier-formed leachate in the southern and central areas has most probably migrated beyond the Wairau Taylor Fan aquifer boundary only a further 0.5 km to the north.

The complex history of landfilling over a 20-year period, the fact that leachate is appearing in deeper (> 25 m) wells and the generally high transmissivities of the gravels forming the Taylor Fan surface, together make interpretation of the leachate migration extremely difficult.

This is further complicated by the presence of the older Brayshaw Parks landfill extending virtually to the mapped southern margin of the Wairau Aquifer, and the preferred interpretation of the present chemical plume is that mixing of Wairau and Taylor Fan water is occurring as far south as Page Street. The effect of this "zone of mixing" on the lower Taylor Fan for some 400-500 m above the mapped Wairau-Taylor Fan boundary is to rapidly dilute leachate and natural Taylor derived waters, at the same time insuring that neither landfill is significantly impacting groundwater resources at least to a depth of 25 m.

Flows through the Wairau Aquifer are considerable greater than those on the Taylor Fan with transmissivities several order of magnitude greater and substantial recharge occurs from the Wairau river itself. In comparison, flows both on and through the Taylor Fan gravels are intermittent due to lower rainfall recharge from the catchment to the south, whilst the flow contribution from the Taylor Pass Landfill itself has been shown to be relatively minor (only 15000-17000 m^3 per annum from direct infiltration). It can therefore be concluded that further substantial expenditure on capping of the Taylor Pass Landfill is unnecessary, although removal of offal discharge to the secure Blue Gums Landfill should still be a priority. Given that leachate from both the Taylor Pass or Brayshaw Park landfill sites is almost certainly being rapidly attenuated by Wairau-derived groundwater without apparent detriment to down-gradient users, revegetation of the Taylor Pass Landfill surface would appear to be adequate to minimise future infiltration. At the same time a deep cut-off to intercept groundwater entering the southern end of the landfill could also be considered to further reduce leachate generation.

Chapter 6

Conclusions and Recommendations

6.1 Project Objectives

The primary objectives of the thesis have been:

- to develop a site hydrogeological model by means of engineering geological mapping, trench logging, geophysical methods, logging of new wells, and correlation of existing bore hole data;

- to characterise the various site cover materials in terms of relevant hydraulic parameters, including the area already capped at the Taylor Pass Landfill site;

- to determine the extent of leachate generation at the Taylor Pass Landfill from shallow groundwater sources derived principally from the Taylor Fan to enable comparison with direct precipitation-generated sources;

- to determine the nature, extent and rate of migration of the leachate plume formed down-gradient from the main Taylor Pass Landfill, including the establishment of a monitoring short term monitoring programme; and

- to assess the influence (if any) of the Taylor Pass Landfill and the older landfill site (Brayshaw Park) on groundwater quality in the down-gradient areas, including possible hydraulic connection with the regional aquifer systems.

6.2 Investigation Methodology

The following investigation methods were implemented in the hydrogeological and hydrogeochemical examination and analysis of the Taylor Pass Landfill, Blenheim:

- Logging of new boreholes
- Correlation of existing lithological well logs
- Landfill site mapping
- Landfill trench excavation and logging
- Preliminary geophysical surveys
- Extrapolation of existing hydrogeological data
- Application of the Water Balance Method for the analysis of precipitation infiltration into a landfill
- Analysis of existing hydrogeochemical data
- Implementation of a intensive project specific geochemical monitoring program
- Laboratory testing of cover materials

Of these methods, primary emphasis has been placed on hydrogeological evaluation and groundwater chemistry. Geophysical method such as TEM and resistivity proved unsatisfactory, and the absence of monitoring wells in the Taylor Pass Landfill meant that data had to be extrapolated from nearby monitoring wells and test pits excavated at the margins of the refuse dump. The short-term monitoring programme, with 6 additional bores including 2 deep ones, allowed for clearer definition of probable leachate extent and migration. However, a major constraint to the interpretations and conclusions from this study have been the relatively saline nature of the Taylor Pass Groundwaters upgrading from the Taylor Pass landfill, and the dominant influence of much higher quality Wairau-derived groundwaters in the vicinity of the Brayshaw Park Landfill.

6.3 Site Hydrogeological Model

The Taylor Pass Landfill is located adjacent to the Taylor River within the modern river gravels of the Taylor Fan. Surficial deposits of the Taylor Fan comprise the Speargrass Formation Lobe on the western side of the Taylor Valley, and the younger Rapaura lobe on the eastern margin in which the Taylor Pass Landfill is situated. The Rapaura lobe of the Taylor Fan has formed by down-cutting and erosion and consequent aggradation of Speargrass deposits since the end of the last glaciation (*circa.* 14 000 years B.P.), whilst undifferentiated older fan gravels underlie the Speargrass and Rapaura deposits. Analysis of borehole logs has not clearly differentiated the younger Rapaura deposits from those of the older underlying units, but the Rapaura fan sequence is thought to be typically 10-20 m thick.

Largely discontinuous alluvial fan deposits comprising highly permeable channel deposits to relatively impermeable silt and clay rich overbank flood deposits form the base of the landfill site and characterise the geology of the Taylor Fan. A fine-grained low permeability "blue pug" layer is known to underlie the southern portion of the Taylor Pass Landfill and may act to perch groundwater within the base of the landfill at its southern end. The water table certainly intercepts the base of the Taylor Pass landfill at the southern end, and projected piezometric levels suggest the water table drops to within 1.5 m of the base through the mid section; an increasing hydraulic gradient in the northern section causes a further drop in water level to 3-4 m below the landfill base in the northern section. Effects of mounding of the water table beneath the landfill remain unquantified, whilst fine-grained relatively impermeable layers similar to the "blue pug" layer are likely to exist along the eastern margin of the landfill where it abuts Rapaura gravels and may account for the presence of seepage in trenches above the level of the projected groundwater table.

Moving away from the landfill, deposits are highly discontinuous due to the nature of the alluvial fan system, and estimated hydraulic conductivities of the order of 5×10^{-4} m/s are considered more representative of channel deposits rather than the laterally extensive fine-grained materials encountered in the monitoring well logs. Correlation of fine-grained zones between wells on the Taylor Fan surface is poor due the nature of their deposition, and it has not been possible to actually define the location of preferential flow paths comprising channel deposits by any of the methods used in this investigation.

The piezometric surface in the Taylor Fan defined by a small-scale survey, has hydraulic gradient of 1/180 near the southern end of the Taylor Pass Landfill, steepening to approximately 1/50 about 400 m north of the landfill. This is constrained by wells at distances up to 300 m from the Taylor Pass landfill, and by data from shallow trenches and bores adjacent to the landfill. It is clear from these contours that the Wairau-derived groundwaters exert a major control on the system in the down-gradient area, with a probable zone of mixing for some 500 m above New Renwick and Alabama Roads where the boundary with the Taylor Fan is traditionally placed.

6.4 Landfill Cover Materials

The Taylor Pass Landfill is covered to variable depths up to 2.1 m with a variety of reconstituted cover soils. The southern end of the landfill is capped with sandy clayey silt and gravel with laboratory permeabilities of the order of 1×10^{-8} to 2×10^{-7} m/s, which are less than resource consent requirements of 1×10^{-7} m/s. The majority of the mid section of the landfill is covered with stockpiled clean fill materials with permeabilities ranging from 1×10^{-8} to 1×10^{-6} m/s, reflecting the variability of disposed clean fill materials. The remainder of the landfill soil cover does not comply with resource consent requirements, but other studies have shown that the permeability of cover soils is not a governing factor in the potential infiltration in such a dry climate.

In order to improve the efficiency of the cover materials vegetative cover such as pine and wattle trees, which currently cover approximately 30% of the landfill, must be promoted to encourage evapotransipiration across the landfill area. The currently operational offal pits not only create a pathway for immediate infiltration into the landfill but also expose a significant amount of surrounding refuse to precipitation effects. Water balance studies suggest that precipitation induced infiltration amounts to approximately 15000-16000 m^3 per annum into the Taylor Pass Landfill, and the present study concludes that revegetation of the site and removal of the offal waste dumping to the secure Blue Gums Landfill site are preferred options for remediation.

6.5 Origin of Leachate

Establishing the extent of infiltration of surface and groundwaters into the refuse body was the main aim of Taylor Pass Landfill site-specific hydrological and hydrogeological investigations. Based on the Water Balance Method (Thornthwaite and Mather, 1955; Fenn *et al.*, 1975) it is expected that some 15000-17000 m^3 of water enters the landfill site from meteoric sources. Infiltration of precipitation is expected to occur over July and August only, following saturation of cover soils during the winter season but under abnormally wet conditions this could occur at other times of the year also. High intensity short duration rainfall events outside of the winter season are not likely to contribute to infiltration. The moisture storage capacity of the cover soils is thought to be the determining factor for percolation rates under the given conditions at the Taylor Pass Landfill, as there is no direct surface water run-on of extraneous waters at the site.

Groundwater has been identified as entering the landfill site on the basis of interpolation of piezometric data (Section 6.3), and from trenching around the perimeter of the landfill in Section 4.5.3. Results of the analysis of groundwater infiltration indicate that in the southern portion of the landfill the water table fluctuates only by about 0.5 m, and it is likely that groundwater intercepts the landfill base perennially, contributing 22000 m^3 annually to the generation of leachate. Seepage at the landfill perimeter observed above the projected water table during landfill trenching also indicates that water is present at least periodically within the mid section of the landfill and indirect evidence suggests groundwater mounding beneath the landfill. In the northern section, annual groundwater level fluctuation of up to 1 m are not sufficient for the water table to intercept the base of the landfill even with substantial mounding.

In addition, since refuse disposal commenced at the northern end of the landfill and progressed southwards, the leachate currently being generated at the southern end of the site is likely to produce leachate of higher toxicity to leachate currently being generated in the older southern end. However as the base of the southern end of the landfill is being regularly flushed with groundwater, the toxicity of leachate being generated is likely to decline rapidly.

In summary then:

- In the southern portion of the landfill 3300 m^3 of meteoric water and 22000 m^3 contribute to leachate generation annually over an area of 5 ha.
- Over the entire landfill then the
- Any infiltration of groundwater in the mid section of the landfill puts the minimum total groundwater and precipitation contribution the leachate generation in excess of 35000 m^3.
- No groundwater is expected to intercept the base of the landfill in the northern part of the landfill.
- Leachate being generated at the southern end of the landfill

6.6 Monitoring Programme and Plume Delineation

A short term monitoring programme devised for this project was aimed at acquiring adequate control on background groundwater quality, assessing deep (>15 m) migration of leachate given that the standard Connell Wagner monitoring program concentrated on shallow (<15m) down-gradient wells and on generating sufficient data within a twelve-month period to adequately define any seasonal trends that may be present. Six additional monitoring bores were installed. Tested parameters were based on major ions and general parameters indicative of landfill leachate and project specific sampling was carried out at approximately two-month intervals over the period from M ay 1999 to March 2000. Additional water chemistry data was also used.

Analysis of results has resulted in the following parameters being identified as those that *may* be considered indicators of leachate contamination in the Taylor Pass Area and it recommended that these parameters be used as regular testing parameters in the future.

Ammonia-N	TOC	Sodium
Bicarbonate	Chlouride	Arsenic
Sulphate	Alkalinity	Manganese
Potassium	COD	Conductivity

Effects of leachate contamination down-gradient of the Taylor Pass Landfill are masked significantly by elevated background salinity in Taylor Pass groundwater likely attributable to high average evaporation and low rainfall. Based on the parameters listed above however, a leachate plume has been detected to at least 800 m down gradient of the Taylor Pass Landfill.

Eastward migration indicates the presence of slow moving a LNAPL plume. Elevated COD, potassium, manganese and COD have been attributed to landfill sources. Westward migration of contamination is controlled in the Taylor River bed where only elevated manganese is detected. Anomalous alkaline waters (up to pH = 12) at Aerodrome Road on the approximately 700 m west of the Taylor Pass Landfill indicate that P28/W3390 may be contaminated from an alternative source associated with activity at the Omaka Aerodrome. It does not appear hydrogeologically feasible that contamination of the Aerodrome well is due to migration from the Landfill and further work in assessing the true source of contamination at Aerodrome Road is recommended.

Although high salinity of background monitoring bores greatly masks contamination down-gradient of the Taylor Pass Landfill. Contamination is indicated in shallow wells by elevated chloride, COD, ammonia, and arsenic. Deep well P28/W2289 at 800 m down gradient of the landfill and offset slightly to the east indicates slight increasing levels of ammonia and manganese and alkalinity and is interpreted as indicating progressive lateral spread of the leachate plume. There is no control over the vertical extent of leachate migration beyond 25 m.

6.7 Possible Impacts on Down Gradient Groundwater Quality

The minimum possible migration distance by hydrogeological parameters is 4 km, which indicates, in contrast to hydrogeochemical evidence, that the leachate plume emanating form the Taylor Pass Landfill is most likely to have already reached the Wairau Aquifer. The lack of indicators in wells on New Renwick Road and the rapid decrease in contaminant concentrations immediately south of New Renwick Road indicates that the plume is likely dispersed and diluted as it moves through a mixing zone extending north from 1 km down-gradient of the Taylor Pass Landfill.

Mixing occurs as Taylor Fan-derived waters merge with the more dominant Wairau Plains groundwater system, which has substantially greater volumes of flow and acts to dilute both the natural saline waters of the Taylor Fan and that groundwater contaminated by Taylor Pass Landfill leachate, until the effects of both the landfill and the natural conditions of Taylor Pass groundwater are essentially undetectable northwards from New Renwick Road. Although there is currently no constraint over the vertical extent of migration, leachate affected groundwater from the Taylor Pass area is certainly being mixed with Wairau groundwater at the same level as the Eltham Road town supply bore and is likely to be mixing at greater depths also.

Under the current conditions however, the leachate from Taylor Pass Landfill does not appear have significant adverse effects on down-gradient water users however with younger leachate continuing to be produced at the site continued rigorous monitoring of both groundwater up and down-gradient of the Taylor Pass Landfill and also within the landfill itself to establish the true level of leachate within the landfill is recommended.

6.8 Recommended Further Work

Based on the conclusions of this project, the following work is recommended:

- Installation of monitoring bores within the landfill to accurately determine the volume of groundwater passing through the mid and southern sections of the landfill.

- Continued monitoring of down-gradient and control bores with a possible change in monitoring parameters to include those parameters indicative of leachate contamination in all monitoring rounds.

- Investigation of possible remedial works to reduce both groundwater and precipitation infiltration into the Taylor Pass Landfill. It is unlikely that these need include full capping as is required by resource consent, rather, additional top soil and vegetation cover will dramatically reduce the amount of precipitation infiltration

- In addition to further work at the Taylor Pass Landfill it is highly recommended that further work be carried out on the western side of the Taylor River in the Aerodrome Road area to establish the origin of contamination of well P28/W3390. It is recommended that this work be carried out separate from continued investigation and monitoring of the Taylor Pass Landfill.

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